EVALUATION OF RAINWATER RETENTION IN PHASED HYDROLOGIC RESTORATION OF THE ROTENBERGER WILDLIFE MANAGEMENT AREA, OCTOBER 1999 - SEPTEMBER 2000.

Stephen M. Smith, Jennifer Leeds, Brian Garrett, and Malak Darwish

Everglades Department, Watershed Research & Planning Division, South Florida Water Management District, 3301 Gun Club Road, West Palm Beach, Florida 33406.

Table of contents

Introduction

The Rotenberger Wildlife Management Area (RWMA) is a degraded wetland in the northern Florida Everglades that will undergo hydrologic restoration in accordance with the Everglades Forever Act and Consent Decree between the Federal and State of Florida governments. Since being isolated from natural sheet flow by the construction of perimeter levees and canals, surface water in the RWMA has for decades originated only from direct rainfall. This limited input can only maintain standing water in the marsh for very short periods of time. Accordingly, the area has been targeted to receive discharge from Stormwater Treatment Area 5 (STA-5) in the year 2000. STA-5 is a constructed wetland designed to reduce nutrient concentrations in upstream waters (primarily agricultural runoff) before supplying them to the marsh to augment natural precipitation.

In accordance with the Army Corps of Engineers 404 permit for STA-5 construction and operation and Memorandum of Agreement among the Florida Fish and Wildlife Conservation Commission, South Florida Water Management District, Florida Department of Environmental Protection, and the Board of Trustees of the Internal Improvement Trust Fund, a phased approach to hydrologic restoration was implemented. This included the establishment and assessment of a 1-year period of increased rainwater retention within the RWMA prior to any discharges from STA-5. To accomplish this, drainage culverts along the southern and eastern levees of the RWMA were partially closed in late September, 1999 to limit outflow from the area, thereby raising water depths and extending the hydroperiod.

This report summarizes physical, chemical, and biological data collected during the rainwater retention period (RRP) from October 1999 to September 2000. It should be noted, however, that sampling of water quality and periphyton was severely limited by drought conditions that resulted in an absence of surface water from December 1999 to August 2000. Thus, the RRP could not be evaluated from the standpoint of increased water depths and duration. Additionally, the whole system was undergoing recovery from a severe fire that occurred in May 1999.

Sampling locations within the RWMA are depicted in Figure 1 and complete descriptions of sampling methods are detailed in Smith et al. (1999). Much of the data is presented and discussed in relation to pre- and post-RRP samplings. Any discussion of significant spatial and/or temporal differences is based on ANOVA computations using Tukey's tests for specific comparisons between means ($\alpha = 0.05$).

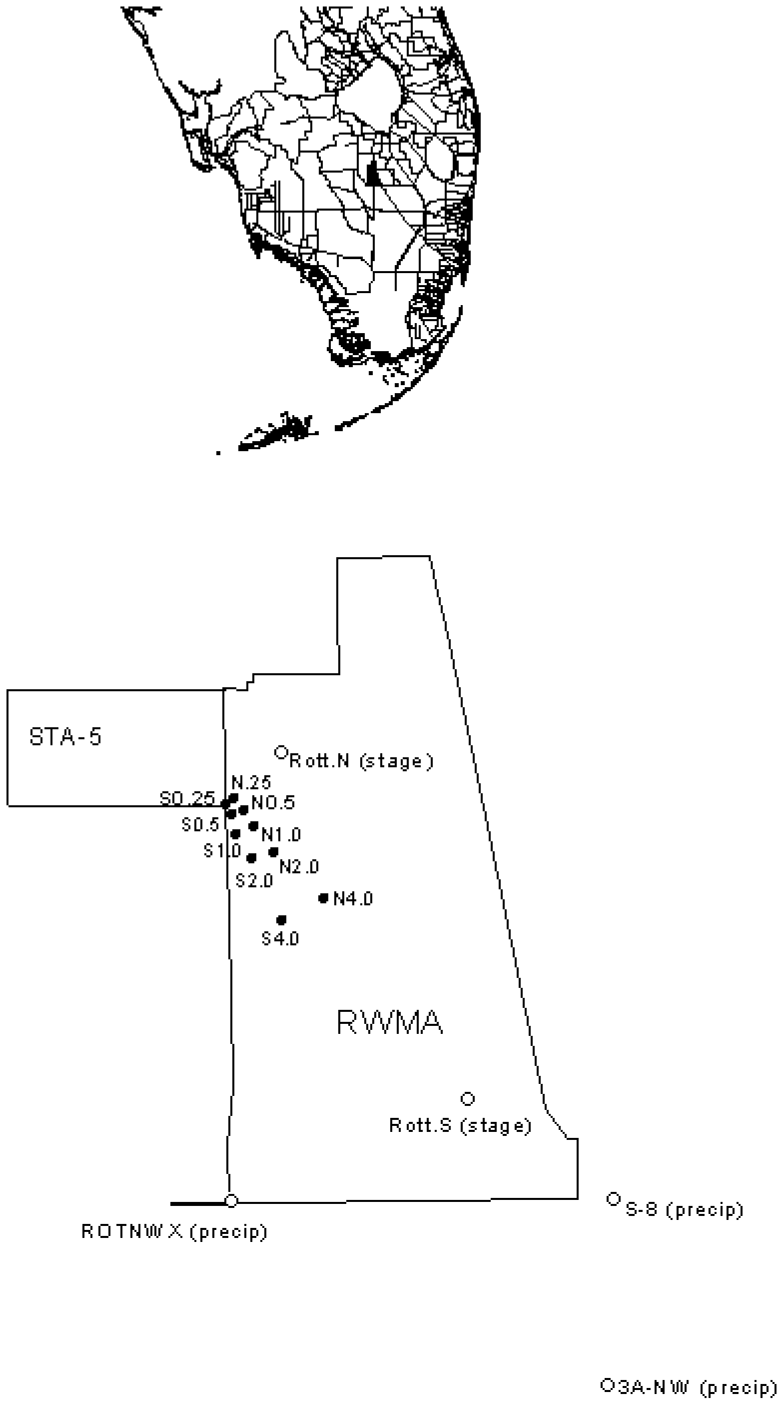


Figure 1. Map showing south Florida (upper map with dark-shaded region representing the RWMA) and the RWMA (lower map) with the locations of STA-5, monitoring stations (solid circles), and stage and precipitation gauges (open circles).

Hydrology

A complete ten-year record from 1990 to present was available from the S-8 weather station situated in the southeast corner of the RWMA (Figure 1). Rainfall during the RRP was 49.9 inches, a value exceeded in every other October - September time period over the last decade with the exception of 1993-1994 and 1996-1997 (Figure 2). Moreover, this value was elevated substantially by a single precipitation event (Hurricane Irene, October 15-16, 1999).

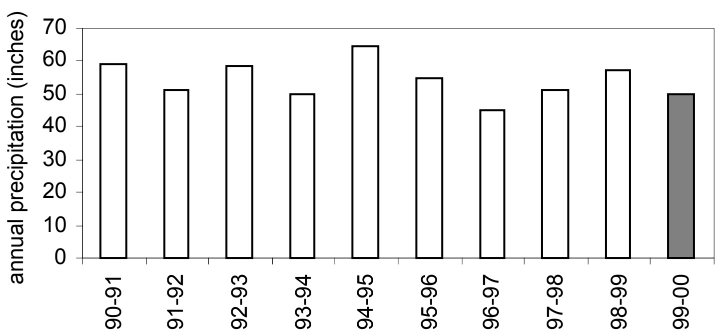


Figure 2. Annual precipitation (October to September time period) over the last ten years as recorded by the S-8 weather station.

Within the RRP exclusively, precipitation data was available from the S-6, S-7, S-8, and 3A-N weather stations for October 1999 through June 2000. Figure 2 summarizes the monthly mean sums of these gauges. For July, data was available only from the ROTNWX weather station. Mean monthly precipitation sums ranged between a minimum of 0.2 inches and a maximum of 9.2 inches in October (Figure 3).

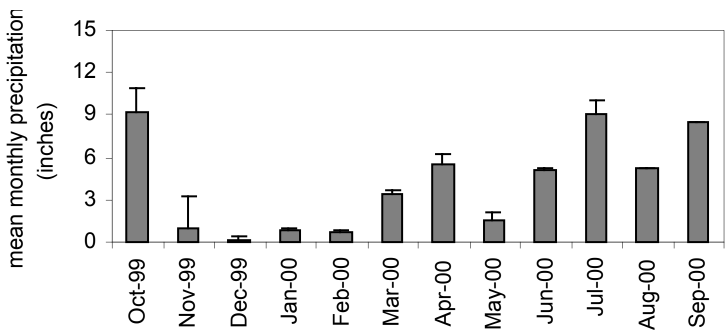


Figure 3. Mean monthly precipitation during the RRP.

Mean instantaneous water depths (calculated from manual readings with a meter stick each month) were highest in October following Hurricane Irene (Figure 4). Subsequently, water levels began to decline rapidly and were below ground by January 2000. Water depths were negative (i.e., below ground) for the rest of the RRP. Readings from two permanent daily stage recorders within the RWMA (Rott.N and Rott.S; Figure 1) document a virtually identical pattern, although only data up to July 2000 was made available at the time of this report (Figure 5).

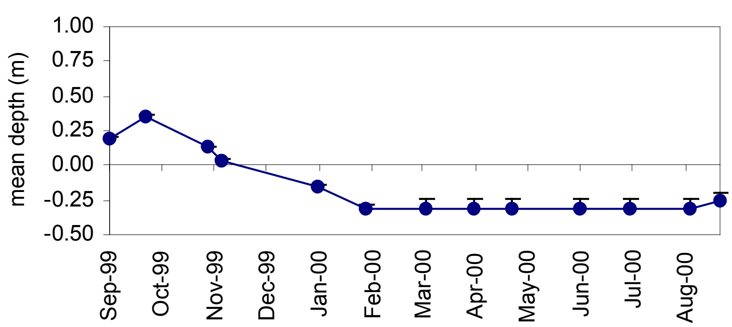


Figure 4. Mean instantaneous water depths (all monitoring stations) during the RRP.

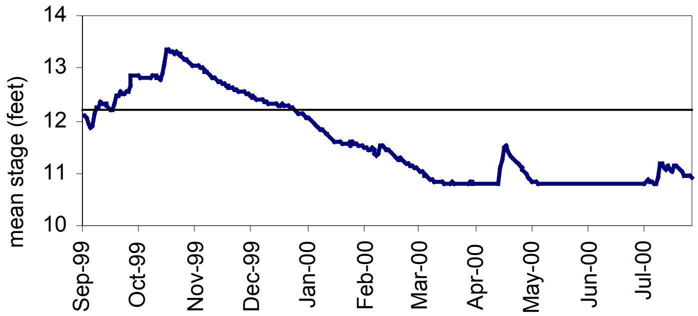


Figure 5. Mean daily stage as recorded by the Rott.N and Rott.S stage recorders during the RRP (horizontal line represents average ground elevation of 12.25 feet above sea level).

<u>Instantaneous surface-water quality</u>

Abbreviations for water quality constituents used in the text and figures are listed in Table 1. Surface water quality samples could only be obtained during October and November of the RRP (1999). During the rest of the year, surface water was either insufficient for sample collection or absent altogether. Given that no well-defined spatial gradient was evident, data were pooled such that the constituent concentrations discussed below are the mean values of all ten monitoring stations. Mean values for November, however, do not include stations $S_{0.25}$, $S_{0.5}$, $S_{1.0}$, and $S_{2.0}$ due to insufficient surface water at the time. The data are presented and compared with pre-RRP samplings conducted in July and September 1999. No samples were collected in August 1999 due to transportation problems to the sampling sites (helicopter unavailability).

Table 1. Surface-water and porewater measurements (* = constituent analyzed for surface-water only).

Constituent	Abbreviation	units
dissolved oxygen	DO	mg/L
conductivity	COND	µhos/cm
pH	pH	pH units
alkalinity	ALK	mg/L
dissolved organic carbon	DOC	mg/L
total kjeldahl nitrogen	TKN	mg/L
total dissolved kjeldahl nitrogen	TKN-F	mg/L
ammonium	NH_3-F	mg/L
nitrate+nitrite	NOx-F	mg/L
nitrite*	NO_2-F	mg/L
total phosphorus	TP	mg/L
total dissolved phosphorus	TP-F	mg/L
soluble reactive phosphorus	PO4	mg/L
total dissolved potassium	K-F	mg/L
total dissolved silica	SiO_2-F	mg/L
total dissolved iron	Fe-F	µg/L
total dissolved magnesium	Mg-F	mg/L
total dissolved zinc*	Zn-F	µg/L
total dissolved calcium	Ca-F	mg/L
total dissolved copper*	Cu-F	µg/L
total dissolved chloride	Cl-F	mg/L
total dissolved sulfate	SO_4-F	mg/L

Alkalinity, conductivity, dissolved oxygen, and pH (Figure 6) - Alkalinity declined from 117.7 mg/L in July to 93.5 mg/L in September, then increased significantly to 139.5 mg/L in October and finally to 195.2 mg/L in November. Conductivity followed a similar pattern, although variations were less pronounced, ranging between 180.6 µhos/cm in September and 246.5 µhos/cm in July. pH averaged 8.03 and 7.95 in July and September, respectively, but decreased significantly to 7.42 in October. pH subsequently exhibited a slight rise to 7.68 in November.

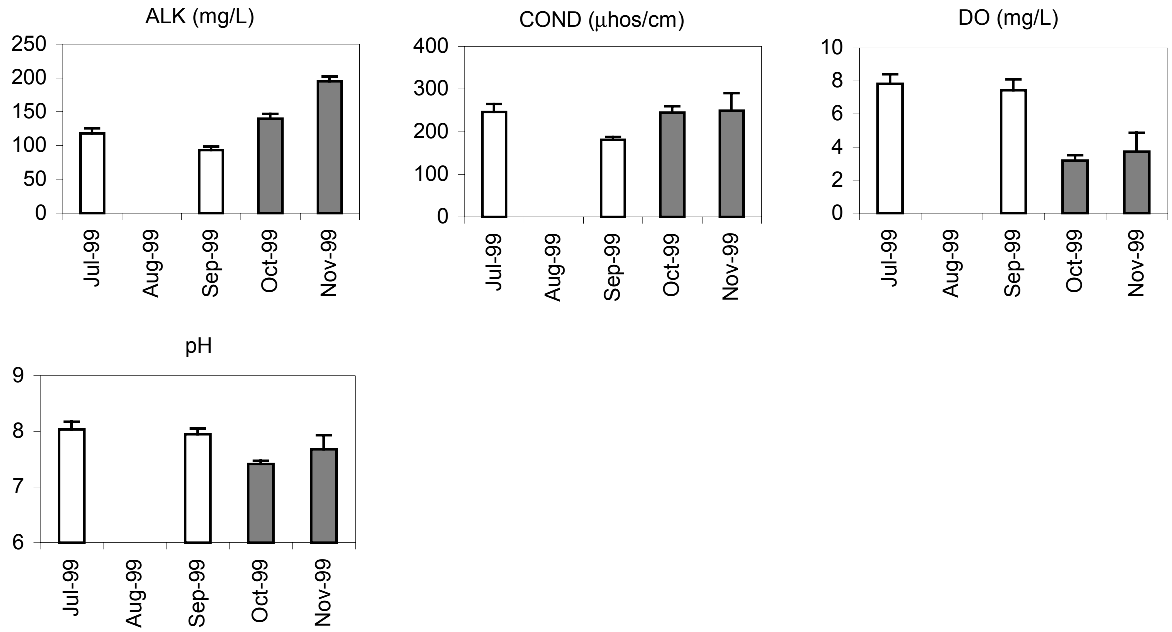


Figure 6. Mean surface-water alkalinity (ALK), conductivity (COND), dissolved oxygen (DO), and pH by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Major nutrients (Figure 7) - NH_3-F concentrations were < 0.01 mg/L in July and September, but increased to 0.73 mg/L by October. By November, a substantial decrease in concentration to 0.31 mg/L had occurred. NO_2-F and NO_x-F followed a similar pattern. For example, NO_2-F was 0.004 and 0.006 mg/L in July and September, respectively; concentrations then increased to 0.039 mg/L in October, subsequently falling to 0.006 by November. NO_x-F concentration increased from 0.004 mg/L in July to 0.011 mg/L in September and 0.034 mg/L in October. By November, however, concentrations had decreased to 0.012 mg/L. TKN and TKN-F concentrations were 3.23 mg/L and 3.13.mg/L, respectively, in July but declined to 1.89 mg/L and 1.56 mg/L, respectively, by September. October and November concentrations ranged between 2.48 and 2.25 mg/L, respectively, for TKN, and between 2.38 and 2.27 mg/L for TKN-F. PO_4-F was undetectable (≤ 0.004 mg/L) in July and October and 0.006 mg/L in

November. TP however, was very high in July with a concentration of 0.082 mg/L, which decreased to 0.047 mg/L by September. October and November TP concentrations were 0.062 and 0.035 mg/L, respectively. TP-F concentration was 0.038 mg/L in July but fell to 0.015 mg/L by November.

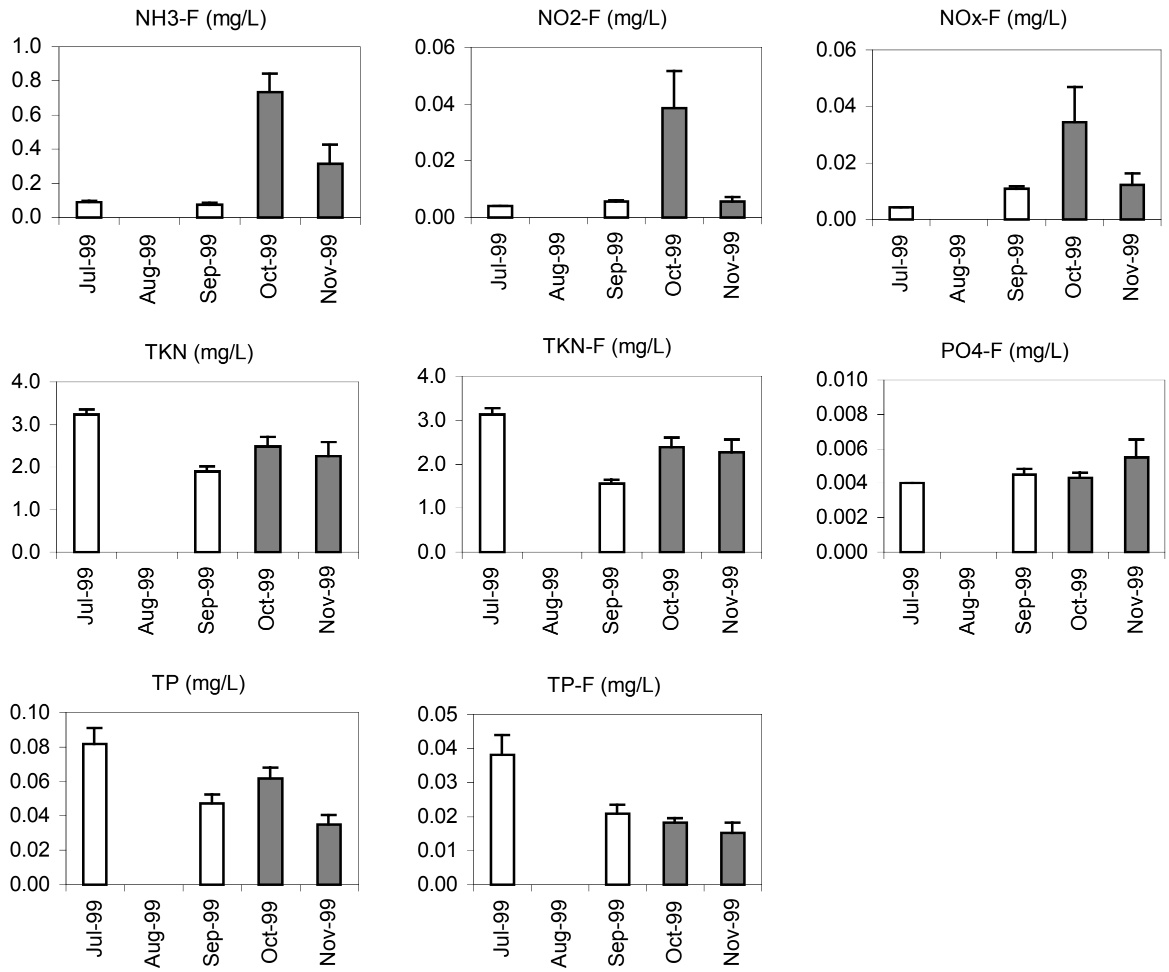


Figure 7. Mean surface-water concentrations of nitrogen (N) and phosphorus (P) species by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Dissolved organic carbon and major ions (Figure 8) - DOC showed a reduction in concentration from 37.0 mg/L in July to 17.9 mg/L in September and then an increase to 27.8 mg/L by November. Cl-F and Na-F declined rapidly from July to September but varied little thereafter. Cl-F concentration decreased from 10.24 mg/L in July to < 2.4 mg/L in September through November. Na-F was 6.89 mg/L in July but < 2.55 mg/L in September through November. Ca-F concentrations were 45.5 and 35.7 mg/L in September and July, respectively, but increased significantly to 53.4 mg/L in October and then to 73.6 mg/L in November. A similar pattern was observed for Mg-F where concentrations ranged between 3.3 mg/L in September and 5.6 mg/L in November. At 2.0 mg/L, K-F concentrations were highest in July. In September to November, however, concentrations were significantly lower (< 1.0 mg/L). SiO_2-F exhibited a continuous reduction in concentration from 3.95 mg/L in July to 0.73 mg/L November. In contrast, SO_4-F concentrations were highly variable, increasing from 3.58 and 3.87 mg/L in July and September, respectively, to 5.38 mg/L in October. SO_4-F concentration then decreased to 0.86 mg/L by November.

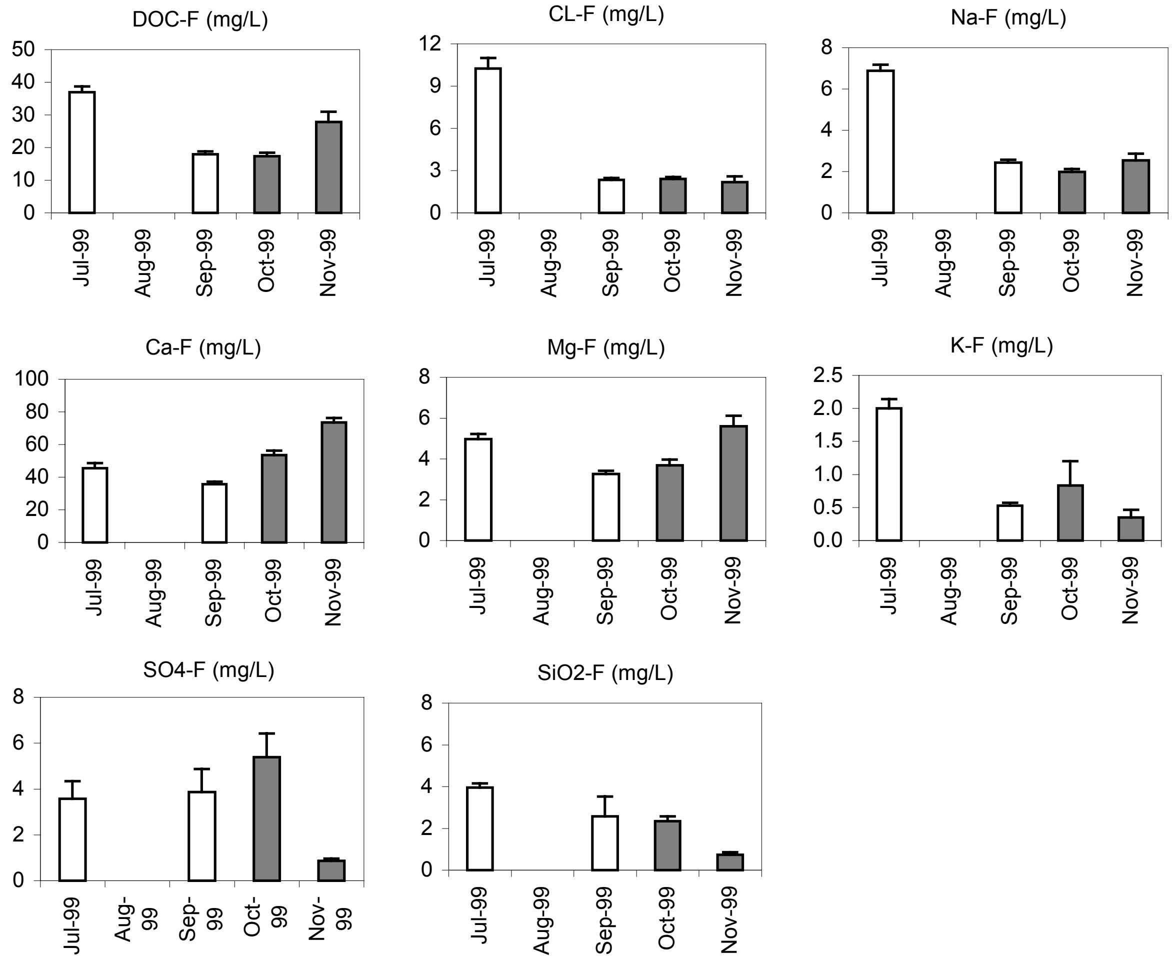


Figure 8. Mean surface-water concentrations of dissolved organic carbon and major ions by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Metals (Figure 9) - Fe-F concentration was 270 μg/L in July, decreasing to 136 μg/L in September. Concentrations rose thereafter to 150 μg/L in October and 434 μg/L in November. Concentration of Zn-F was significantly lower in September (6.0 μg/L) than in July (17.8 μg/L). Zn-F concentration data for subsequent months were discarded due to sample contamination. Cu-F concentrations were generally very low and declined from a high value of 0.53 μg/L in September to 0.34 μg/L in November.

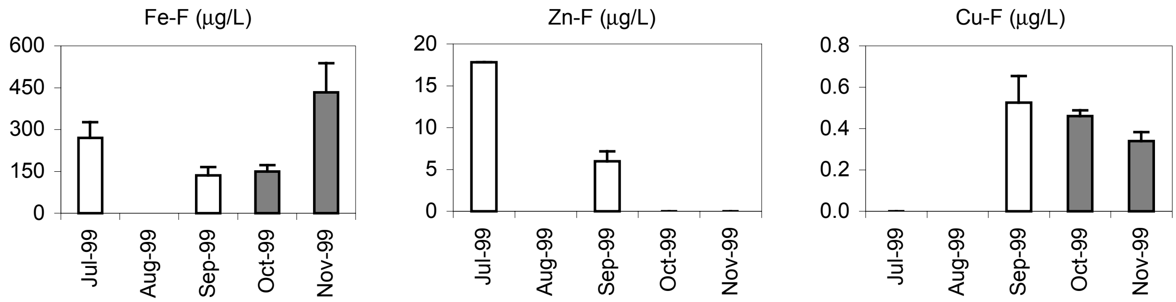


Figure 9. Mean surface-water concentrations of trace metals by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Pore-water quality

Due to low water levels during the RRP, quarterly porewater samples could only be obtained on a single occasion (December 1999). These data are discussed and presented below by comparisons to previous samplings conducted just prior to the RRP. Concentration values are grand means calculated from individual station means that are determined by samples drawn from 2 pore-water wells at each station. Stations $S_{4.0}$ in June and $S_{0.5}$ and $S_{1.0}$ in December are not included in mean values due to insufficient water levels at time of sampling.

pH and dissolved organic carbon (Figure 10) - From June to September, pH decreased from 8.02 to 6.98. In December, pH was 7.29. DOC-F exhibited a decreasing trend over time with concentrations ranging between 85.6 mg/L in June and 50.0 mg/L in December.

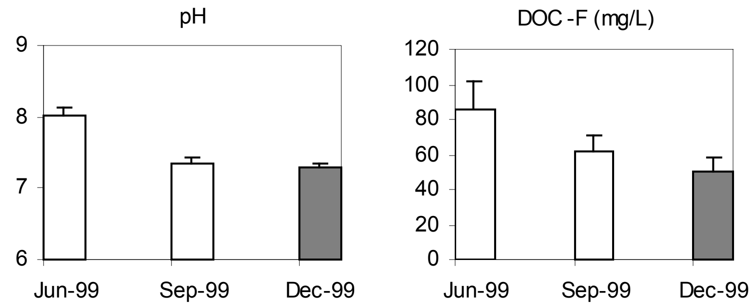


Figure 10. Mean pore-water pH and dissolved organic carbon by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Major nutrients (Figure 11) - NH_3-F concentration was 13.3 mg/L in June but declined to 7.8 mg/L in September and 6.1 mg/L in December. NO_x-F was quite variable with a concentration of 0.2 mg/L in June, rising to1.6 mg/L in September, and declining to 1.0 mg/L in December. TKN-F decreased slightly from a concentration of 16.8 mg/L in June to 12.1 mg/L in September and 10.0 mg/L in December. PO_4-F concentrations ranged between 0.10 in June and 0.05 mg/L in September but increased significantly to 0.19 mg/L in December. TP-F concentrations were 0.094 mg/L in June and 0.093 mg/L in September. A concentration of 0.242 mg/L was measured in December. This increase, however, was not statistically significant due to high variability among stations.

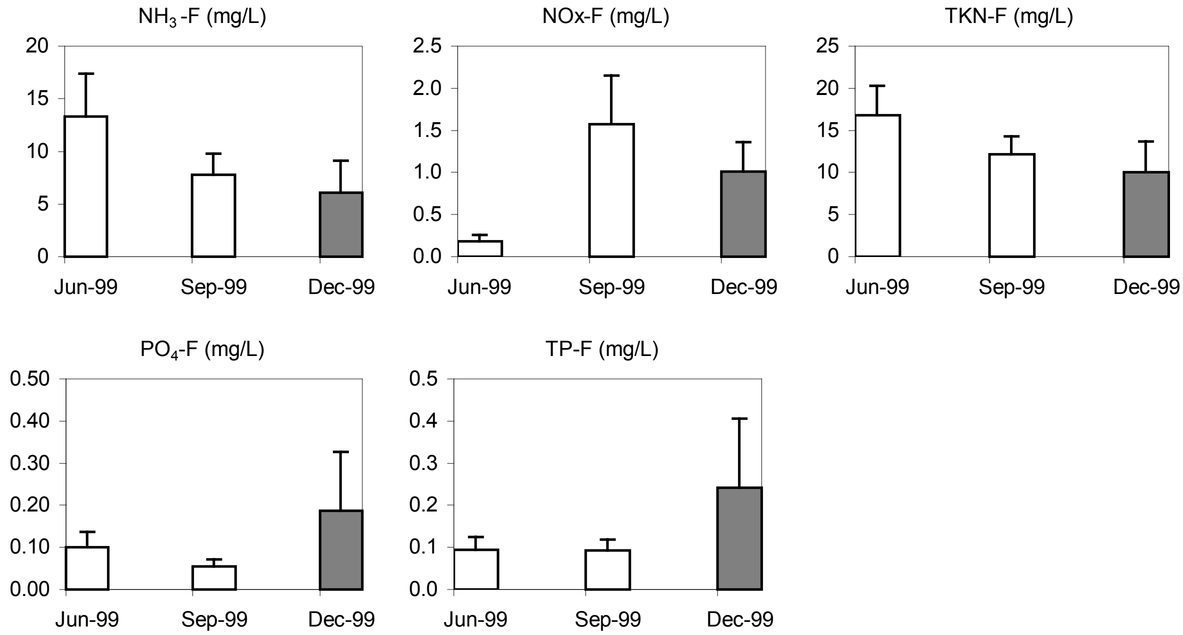


Figure 11. Mean pore-water concentrations of nitrogen (N) and phosphorus (P) species by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Major ions (Figure 12) - Cl-F decreased significantly from a concentration of 30.2 mg/L in June to 12.5 mg/L in September and then to 5.8 mg/L in December. Ca-F concentrations also exhibited a reduction over time, declining from 151.9 mg/L in June to 112.2 mg/L in December. Mg-F concentrations followed a nearly identical trend, falling from a high value of 10.9 mg/L in June to 8.1 mg/L in December. K-F declined more rapidly as concentrations decreased from 4.3 mg/L in June to 2.6 mg/L in September and 1.2 mg/L in December. SO_4-F concentrations in June and September were very similar at 68.2 and 60.1 mg/L, respectively. By December, however, SO_4-F concentration was only 2.4 mg/L. Concentration of S-F decreased from 0.19 mg/L in June to 0.05 mg/L in September but then increased significantly to 0.12 mg/L by December.

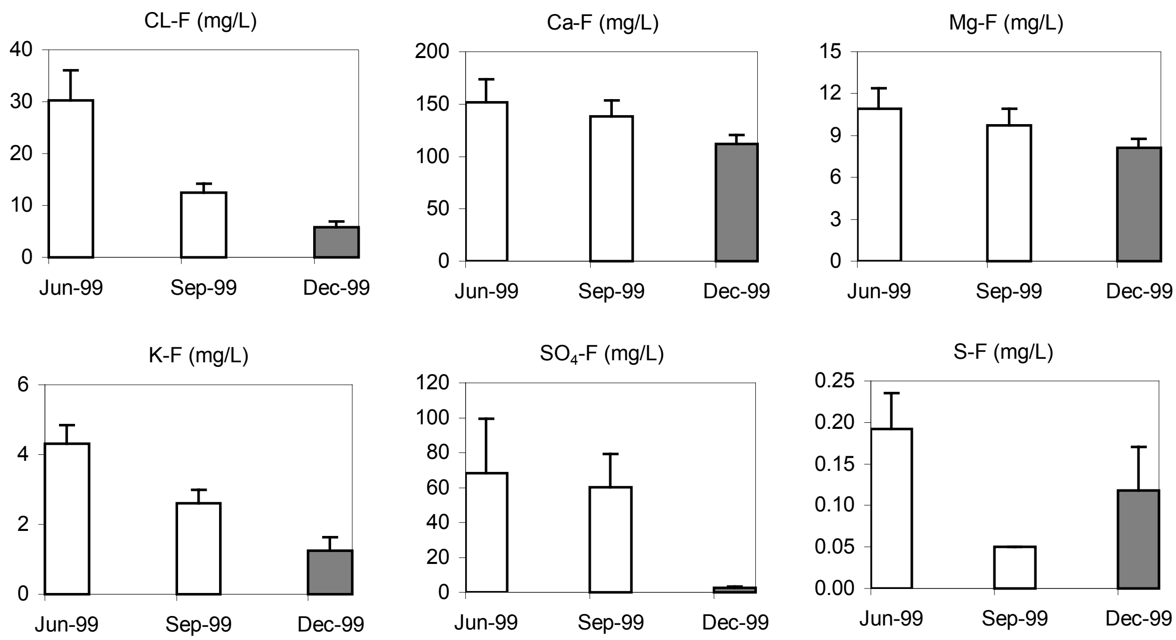


Figure 12. Mean pore-water concentrations of major ions by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Metals (Figure 13) - Fe-F concentrations were high (> 1500 μg/L), but varied little over time.

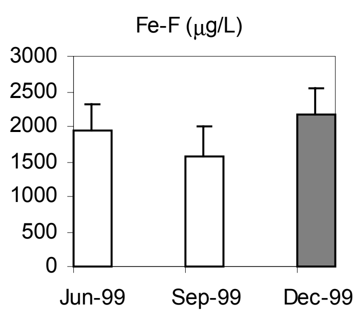


Figure 13. Mean pore-water concentrations of iron by sampling date (open histograms = pre-RRP, shaded histograms = RRP).

Diel surface water quality

Due to prolonged periods of insufficient or absent surface water, no pre-RRP Hydrolab™ data are available for comparison to the RRP. On October 19, 1999, six Hydrolab™ units were deployed at stations $N_{0.25}$, $N_{1.0}$, $N_{4.0}$, $S_{0.25}$, $S_{1.0}$, and $S_{4.0}$ and programmed to record temperature (TEMP), pH, conductivity (COND), and dissolved oxygen (DO) at 30 minute intervals for ~3 days.

Diel TEMP variation was very similar among stations, generally ranging between 24 and 34°C (Figure 14). In contrast, diel pH exhibited considerable differences among stations (Figure 15). In particular, the amplitude of pH fluctuations at stations $N_{1.0}$, $N_{4.0}$, and $S_{4.0}$ was much lower than at $N_{0.25}$, $S_{0.25}$, and $S_{1.0}$. Additionally, the latter stations had pH values exceeding 8.0 whereas all values recorded at the former stations were lower than 8.0. Patterns of diel COND also showed considerable variation among stations (Figure 16). Large fluctuations occurred at $N_{0.25}$, $S_{1.0}$, and, to a lesser extent, $N_{1.0}$. Much smaller fluctuations were observed at $N_{4.0}$ and $S_{4.0}$. In addition, values at these stations were very low (< 250 µhos/cm) compared to all other stations. COND at $S_{1.0}$ fluctuated relatively little until the final day when values steadily increased.

DO concentration exhibited extreme diel fluctuations, the amplitude of which decreased at the most interior stations (i.e., $N_{4.0}$ and $S_{4.0}$) (Figure 17). For example, DO ranged between 0.3 mg/L and 13.7 mg/L at $N_{0.25}$ and between 0.7 mg/L and 6.8 mg/L at $N_{4.0}$. A similar difference in range was observed between $S_{0.25}$ and $S_{4.0}$. In general, nighttime minima were very low with concentrations < 1.0 mg/L at all stations except $S_{1.0}$.

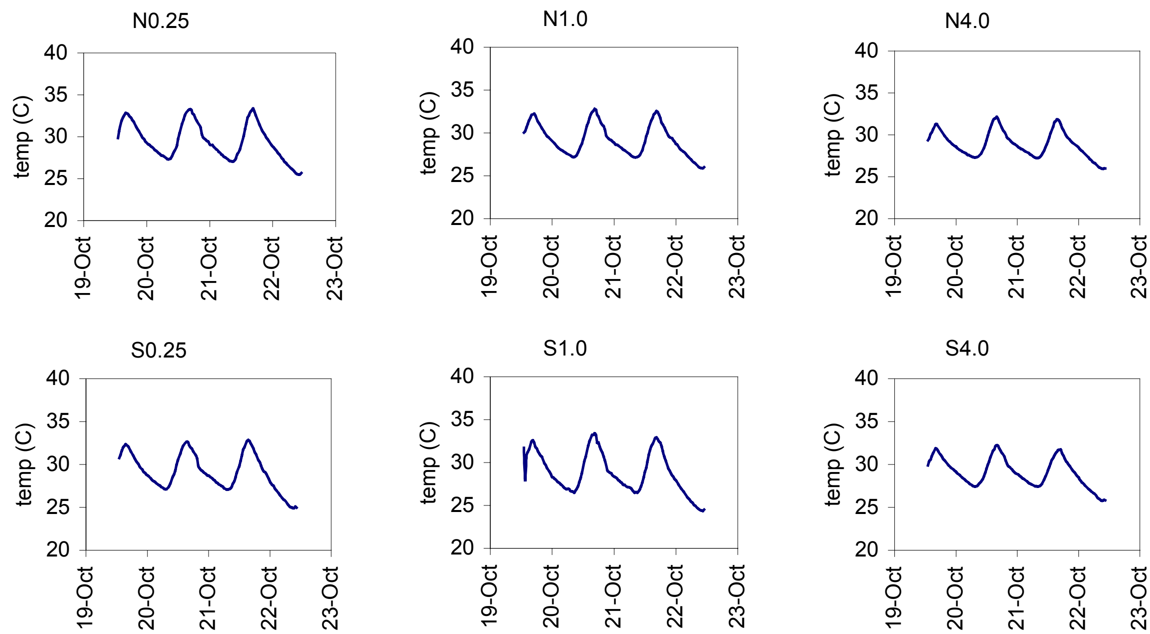


Figure 14. Diel surface-water temperature (°C) during October 19-23, 1999 by station.

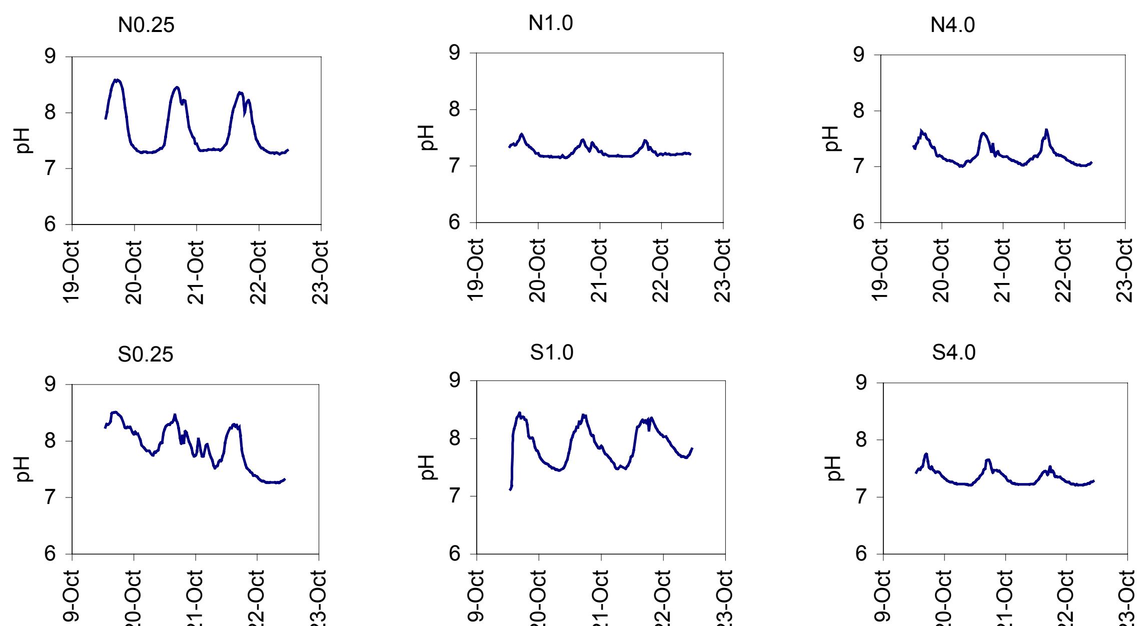


Figure 15. Diel surface-water pH during October 19-23, 1999 by station.

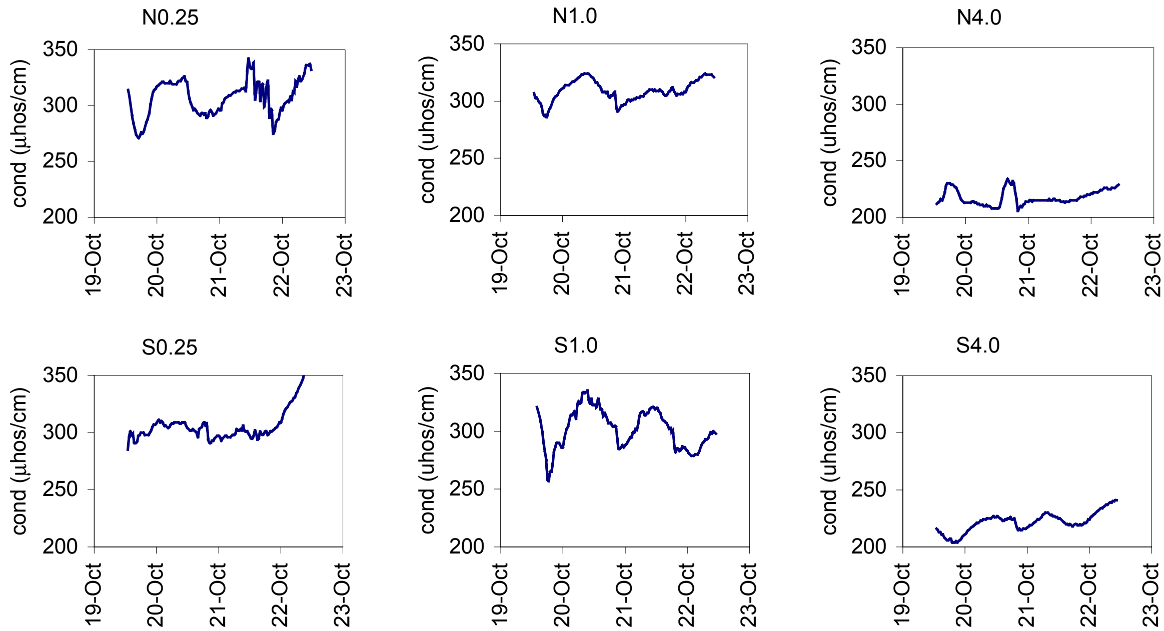


Figure 16. Diel surface-water conductivity (µhos/cm) variation during October 19-23, 1999 by station.

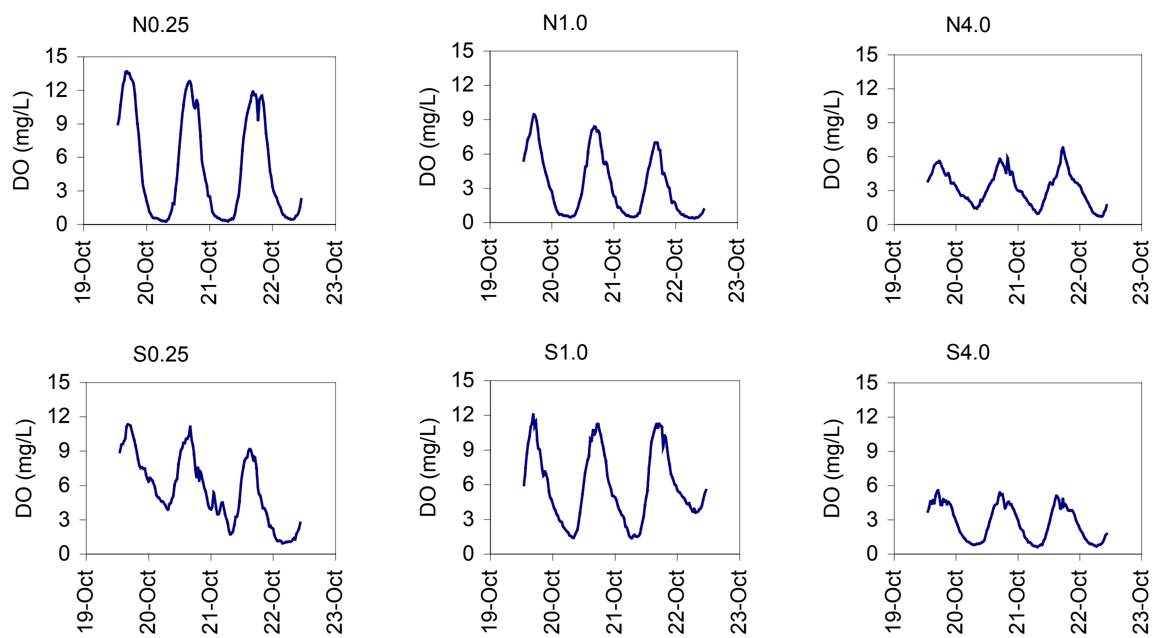


Figure 17. Diel surface-water dissolved oxygen variation during October 19-23, 1999 by station.

Soil properties

Soil samples were collected by coring in September 2000 and sent to DB laboratories for analysis of TC, TN, TP, TCa, bulk density, ash content, and inorganic P in 0-2, 2-10, and 10-20 cm depth increments. At the time of this report, analyses had not been completed. However, properties of soils collected in June 1999 (pre-RRP) are described in Smith et al. (2000) (attachment A).

To date, soil accretion/subsidence has not been measured because of vandalism to established plots. Specifically, the stainless steel rods (approximately 40) that are used to measure sedimentation were removed. Additionally, all of the areas marked by feldspar were trampled by foot.

Periphyton

The development of periphyton communities in RWMA was inhibited by lack of surface water during much of the RRP (see Figure 18). Natural periphyton biomass, productivity, taxonomy, and tissue nutrient concentrations were sampled in August 1999 in adherence to the predetermined monitoring schedule. Although prior to the RRP, these data are thought to be representative of wet season periphyton in subsequent months within the RRP (i.e., October and November). During the following wet season (i.e. August 2000), however, periphyton was non-existent. Biomass estimates from periphytometer-periphyton were obtained in November 1999.

Figure 18. Photograph showing dry conditions at station $S_{1.0}$ during the RRP (March 2000).

Natural periphyton biomass (Figure 19) - Naturally occurring periphyton biomass (ash-free dry weight) was estimated from harvests done within triplicate 0.25 m^2 plots in August 1999. At this time, periphyton was largely present as epipelon, which had accumulated in areas devoid of vegetation from muck-burning in May 1999. By comparison with October 1998, when macrophyte density was much higher, periphyton biomass in August 1999 was relatively high as a result of increased light availability. Epipelon biomass ranged between 0 ($S_{4.0}$) and 115.2 g/m^2 ($N_{2.0}$). Where present, epiphyton and metaphyton biomass was < 15 g/m^2.

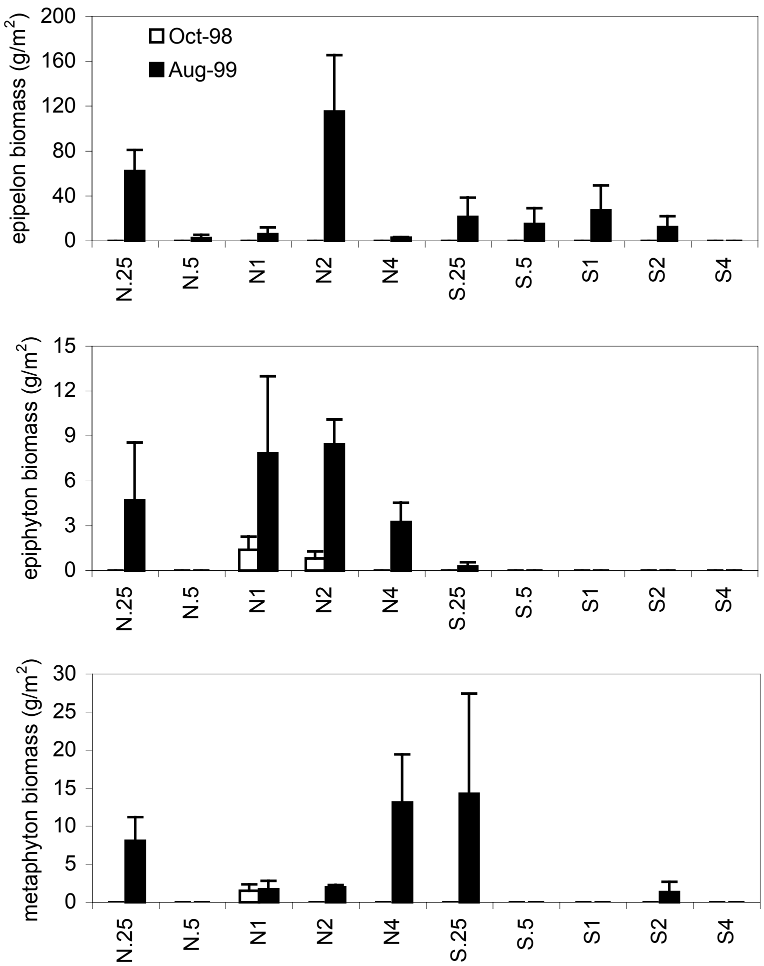


Figure 19. Mean epipelon, epiphyton, and metaphyton biomass (ash-free dry weight) by station in October 1998 (open histograms) and August 1999 (solid histograms).

Primary productivity (Figure 20) - In October 1998, the dominant component of periphyton communities used in the productivity assays was epiphyton. In August 1999 N-transect samples were metaphyton (although this was essentially epipelon that had recently become buoyant enough to float on the surface) while S-transects represented epipelon. Net primary productivity in August 1999 ranged between 0.9 ($N_{0.25}$) and 3.3 $mgO_2/g/\mu mole$ PAR ($S_{1.0}$) (Figure 20) and was very high compared to October 1998.

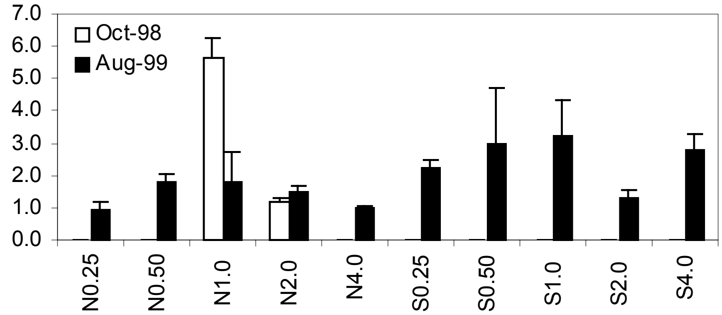


Figure 20. Net periphyton primary productivity by station in October 1998 (open histograms) and August 1999 (solid histograms).

Tissue nutrients (Figure 21) - In October 1998, TP concentrations in the dominant periphyton component (epiphyton) were < 800 mg/kg. In August 1999 concentrations ranged between 1,027 ($S_{4.0}$) and 2,690 mg/kg ($N_{0.5}$) for epipelon and metaphyton. TN concentrations in October 1998 were similar to those in August 1999, which ranged between 11,967 ($S_{2.0}$) and 26,933 mg/kg ($N_{0.25}$). TC exhibited a similar pattern, with concentrations slightly lower in August 1999 than October 1998 (at common stations) and ranging between 129,333 ($S_{2.0}$) and 333,333 ($N_{4.0}$).

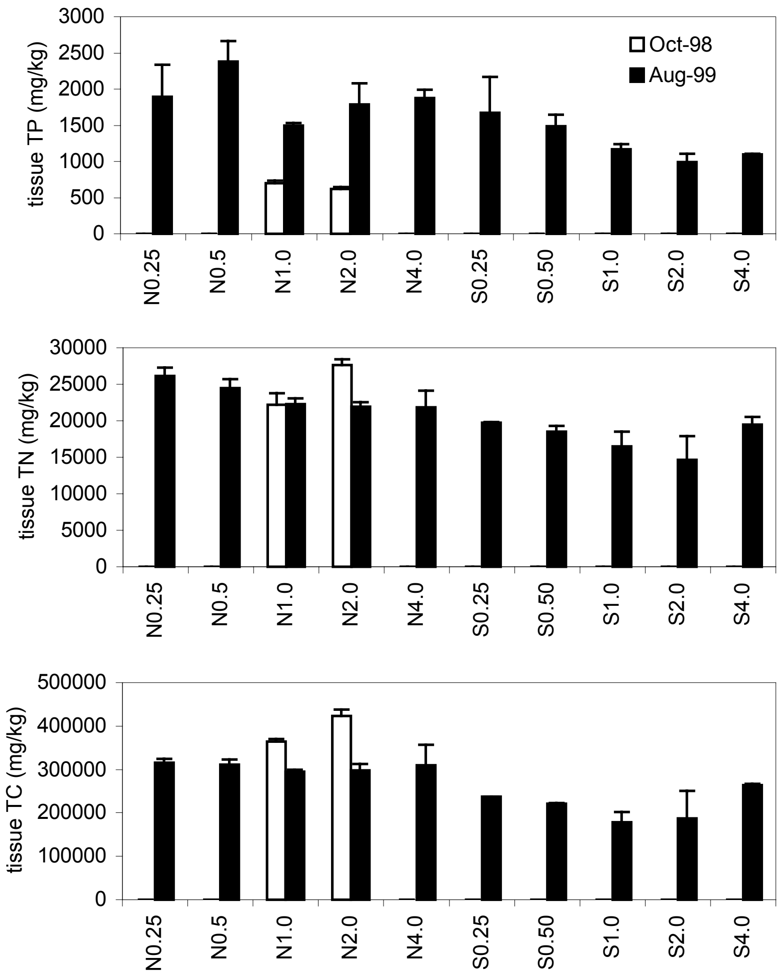


Figure 21. Concentrations of periphyton total phosphorus (TP), total nitrogen (TN), and total carbon (TC) by station in October 1998 (open histograms) and August 1999 (solid histograms).

Periphytometer-periphyton (Figure 22) - During the RRP, periphytometer-periphyton biomass data were obtained from periphytometers floating in the water column for approximately 1 month (October 22 to November 29, 1999). Accumulated biomass over this period was, in general, much higher than that of October 8 to December 17, 1998, even though the latter incubation period was more than twice as long. Among stations, biomass was highly variable, ranging between 1.6 ($N_{4.0}$) and 40.5 g/m^2 ($S_{0.5}$).

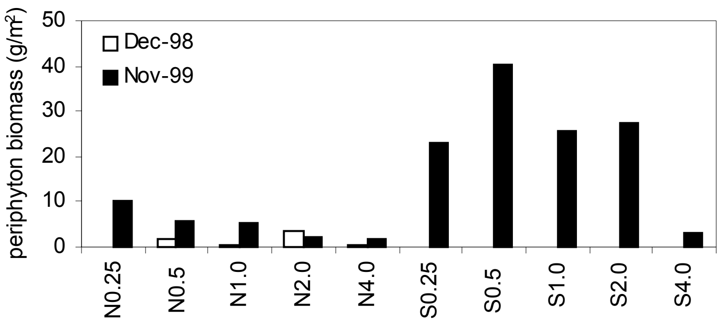


Figure 22. Periphytometer-periphyton biomass by station for time periods ending in December 1998 (open histograms) and November 1999 (solid histograms).

Taxonomy – Taxonomic analyses were conducted by the Florida Department of Environmental Protection on natural periphyton grab samples collected in October 1998 and June 1999 and on periphytometer-periphyton samples from December 1998 and July 1999. Table 2 lists the top ten species (or genera where cells could not be identified to species) ranked by abundance for each sample. These species make up at least 75% of the community by cell density. A number of genera, including *Lyngbya*, *Oscillatoria*, *Microcystis*, *Anabaena*, *Stigeoclonium*, and *Oedogonium* that were found in these samples are considered indicators of eutrophic conditions (Wood and Maynard 1974, Swift and Nicholas 1987, McCormick et al. 1996, McCormick et al. 1998). *Scytonema*, which is prevalent in low nutrient, high mineral water, (Browder et al. 1994), was present in one epiphytic grab sample in October 1998.

Table 2. Top ten genera and species ranked by % abundance in grab and periphytometer samples.

Rank	Oct-98 (Epiphyton)	Jun-99 (Metaphyton)	Jun-99 (Epipelon)
1	*Lyngbya*	*Kentrosphaera*	*Oscillatoria*
2	*Oscillatoria*	*Chlorococcum*	*Anabaena*
3	*Microcystis*	*Oscillatoria*	*Nitzschia palea*
4	*Fortiea bossei*	*Anabaena*	*Oscillatoria limnetica*
5	*Synechococcus*	*Merismopedia minima*	*Merismopedia minima*
6	*Scytonema*	*Lyngbya*	*Chlorococcum*
7	*Aphanothece*	*Nitzschia palea*	*Coelastrum microporum*
8	*Anabaena*	*Merismopedia tenuissim*	*Synechococcus cedrorum*
9	*Chorococcaceae*	*Scenedesmus*	*Scenedesmus acuminatus*
10	*Coelastrum sphaericum*	*Lepocinclis*	*Scenedesmus bijuga*

	Dec-98 (periphytometer)	Jul-99 (periphytometer)
1	*Lyngbya*	*Lyngbya*
2	*Achnanthes*	*Oscillatoria*
3	*Synechococcus*	*Anabaena*
4	*Coleochaete*	*Synechococcus*
5	*Oscillatoria*	*Oscillatoria formosa*
6	*Anabaena*	*Lyngbya aerugineo-caerulea*
7	*Stigeoclonium*	*Oedogonium*
8	*Aphanocapsa*	*Oscillatoria agardhii*
9	*Oedogonium*	*Scenedesmus*
10	*Gloeocystis*	*Coelastrum*

Macrophytes

Biomass - Macrophyte biomass estimates were obtained by manually harvesting above- and below-ground plant matter from triplicate 0.25 m^2 plots within *Cladium jamaicense* (C)- or *Typha domingensis* (T)-dominated stands at the 0.25, 1.0, and 4.0-km stations of both transects (Smith et al. 1999). The March 2000 harvest revealed that substantial regrowth had occurred since the May 1999 fire (Figure 23). *Typha* and *Cladium* live leaf and root biomass was similar to May 1999 estimates. *Cladium* live leaf biomass ranged between 0 ($S_{0.25C}$, $S_{1.0C}$) and 696 g/m^2 ($S_{4.0C}$). Dead leaf biomass was much lower, ranging between 0 ($S_{0.25C}$, $S_{1.0C}$) and 440 g/m^2 ($N_{4.0C}$). Root biomass exhibited a larger range of 79 (dead, remnant sawgrass roots, $S_{0.25C}$) to 1,314 g/m^2 ($N_{4.0C}$).

One essential difference between the April 1998 and May 1999 vs. March 2000 harvests is that *Typha* was present in many *Cladium*-dominated plots where it had been previously absent. Such invasions occurred at stations $N_{0.25C}$, $N_{1.0C}$, $N_{4.0C}$, and $S_{4.0C}$ where *Cladium* had been completely eliminated by muck-burning. In March 2000, *Typha* live leaf biomass was similar to *Cladium*, with a minimum value of 32 ($N_{4.0T}$) and a maximum of 418 g/m^2 ($N_{1.0T}$). In contrast to *Cladium*, *Typha* dead leaf biomass was much higher relative to live leaves, ranging between 106 ($N_{4.0T}$) and 462 g/m^2 ($S_{4.0T}$). *Typha* root biomass in March 2000 was similar to other components but lower than *Cladium* at several stations. Estimates ranged between 28 ($N_{4.0C}$) and 645 g/m^2 ($S_{4.0T}$).

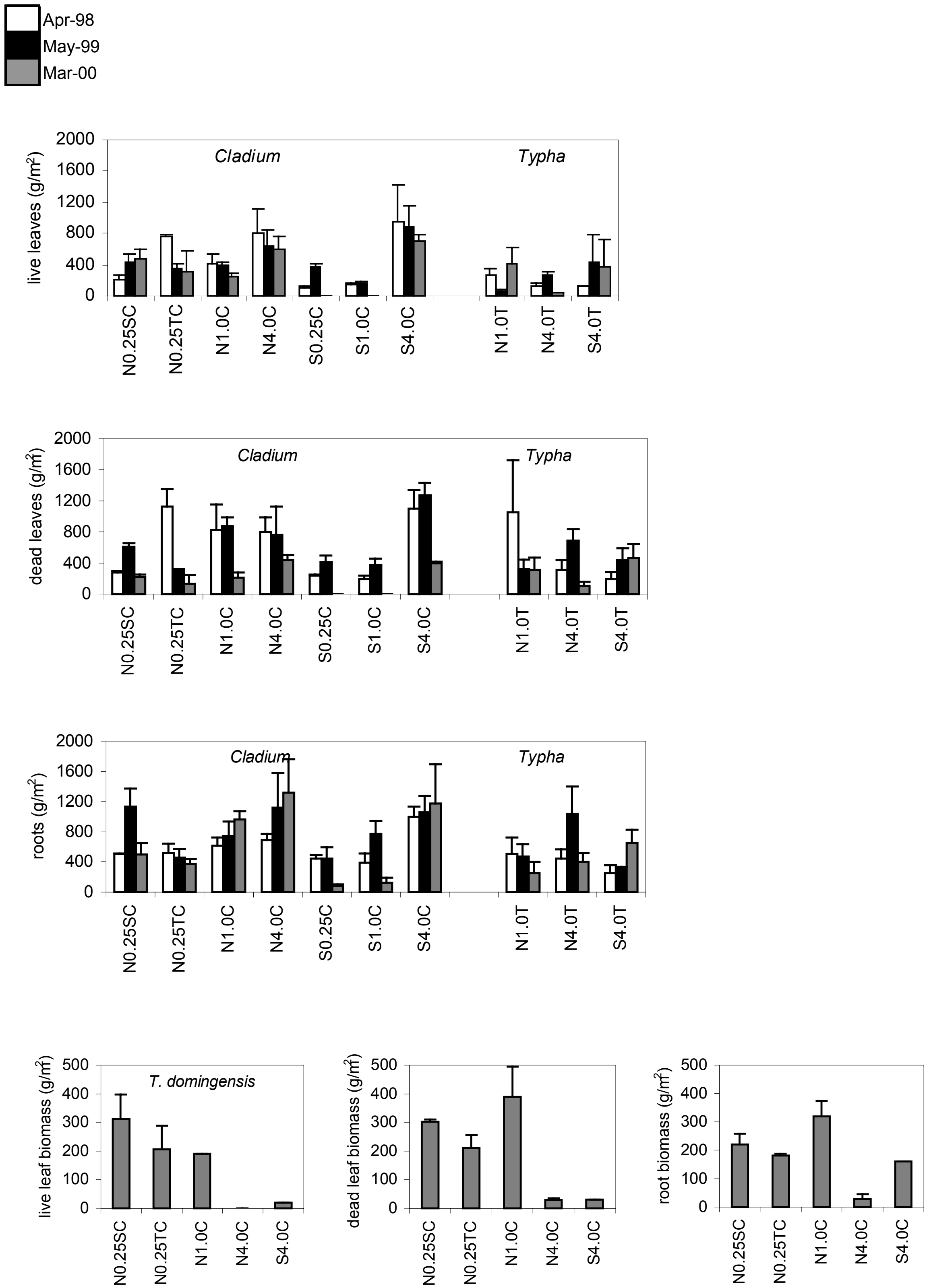


Figure 23. Mean component biomass of *Cladium* and *Typha* within *Cladium* (C)- and *Typha* (T)-designated habitats by station and date.

Non-*Typha* and *Cladium* biomass was highly variable among years and stations but generally lower in *Cladium* habitat compared to *Typha* habitat (Figure 24).

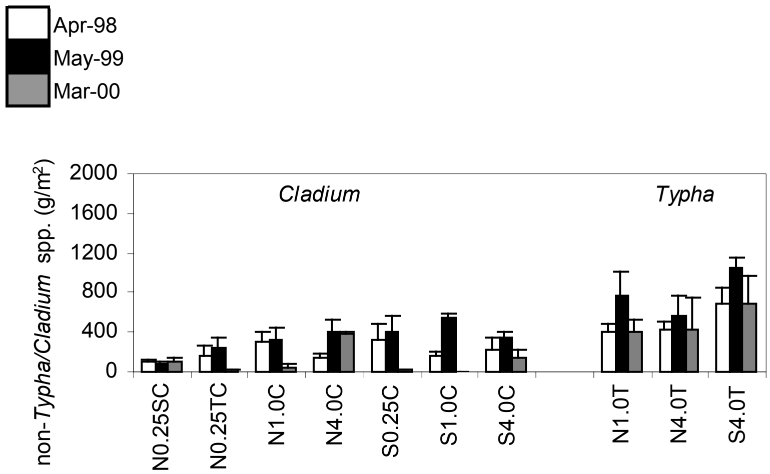


Figure 24. Mean non-*Typha*/*Cladium* biomass within *Cladium* (C)- and Typha (T)-designated habitats by station and date.

Plant densities - Numbers of *Cladium* and *Typha* plants in August 2000 were highly variable among stations and years (Figure 25). *Cladium* ranged between 0 at muck-burned stations ($N_{0.25TC}$, $S_{0.025C}$, $S_{1.0C}$) to 64 plants/m^2 in surface-burned areas ($N_{4.0C}$). At $N_{4.0T}$, $S_{1.0T}$, and $S_{4.0T}$, no live plants of *Typha* were present at the time the survey was conducted as a result of prolonged drought. At $N_{0.25TC}$ and $N_{1.0C}$, however, a few live *Typha* plants were present and enumerated (Figure 26).

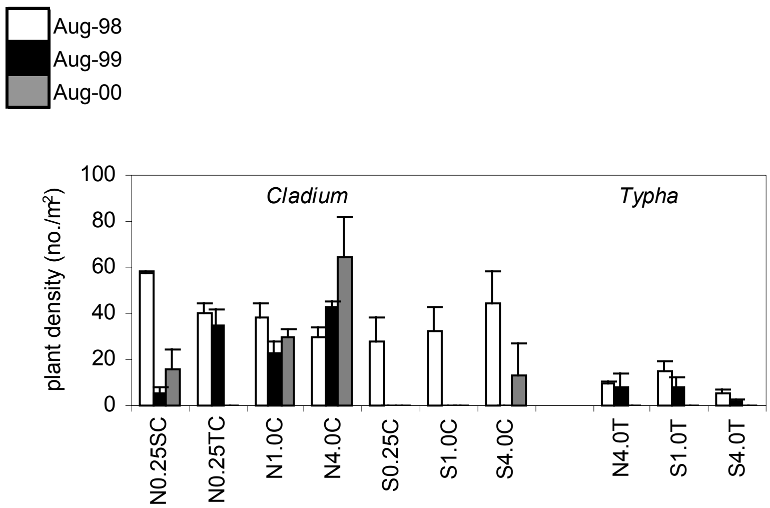


Figure 25. Mean *Cladium* and *Typha* plant density within *Cladium* (C)- and *Typha* (T)-designated habitats by station and date.

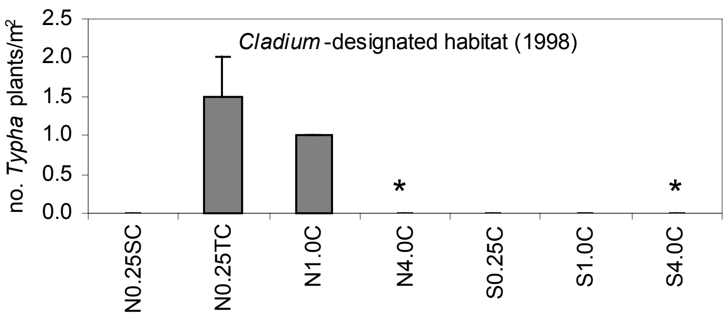


Figure 26. *Typha* plant density in August 2000 in areas originally (1998) designated as *Cladium* habitat (asterisk designates areas where *Typha* has invaded, but all leaves were senescent from drought at the time of enumeration).

Non-*Typha/Cladium* plant densities were generally higher than those of *Typha* or *Cladium* (Figure 27). At many stations, particularly those that were muck-burned, densities decreased significantly from August 1998 to August 1999. By August 2000, however, substantial recovery had occurred and densities at many stations were similar to those in 1998. At $S_{0.25C}$ and $S_{1.0C}$, however, vegetation still had not become established in many of the permanent plots. Species diversity followed the same pattern.

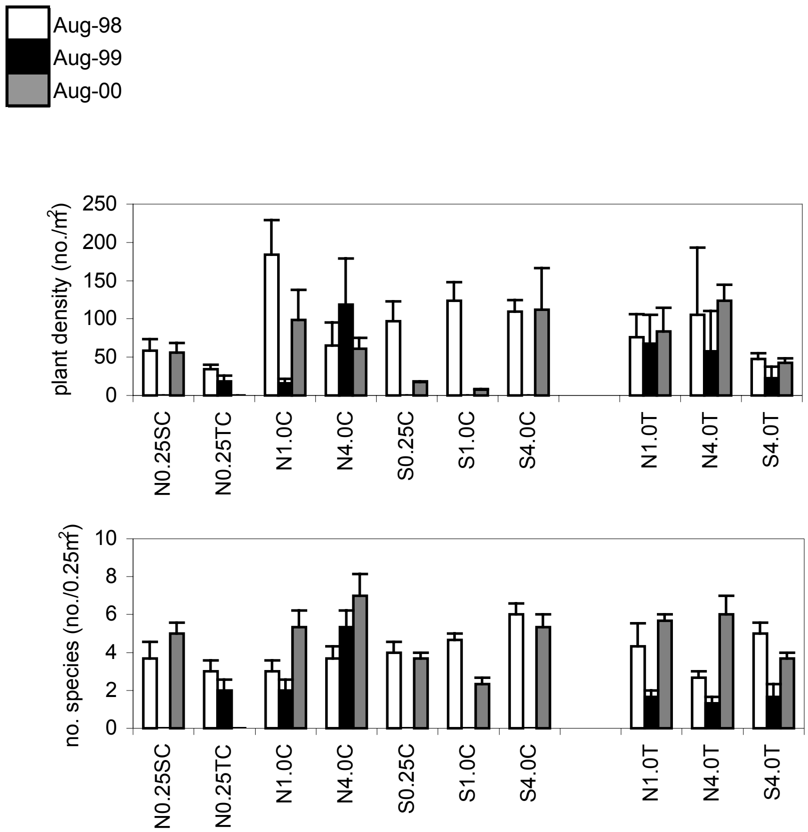


Figure 27. Non-*Typha*/*Cladium* plant density and species richness in *Cladium* (C) and *Typha* (T)-designated habitat in 1998, 1999, and 2000.

Stand heights (Figure 28) - At a number of stations, *Cladium* and *Typha* live leaves were absent due to leaf senescence from drought conditions. Where there were live leaves to measure, heights were significantly higher than August 1999 (post-fire) values but similar to August 1998 (pre-fire) values.

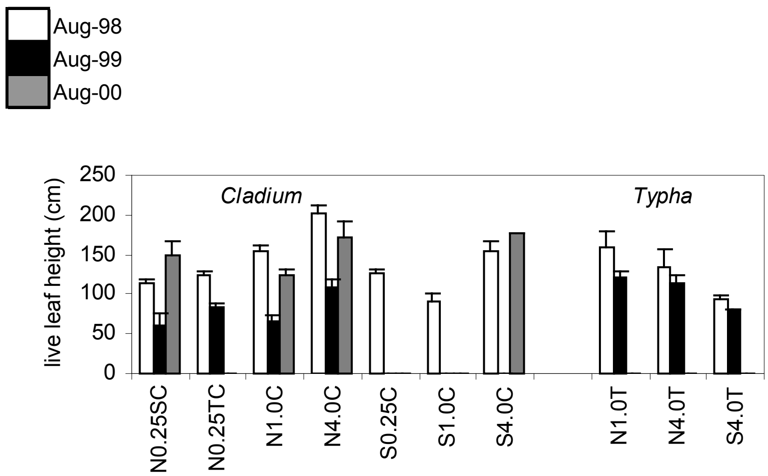


Figure 28. Mean *Cladium* and *Typha* leaf heights within *Cladium* (C)- and *Typha* (T)-designated habitats by station and date.

Tissue nutrients: Cladium TP (Figure 29) - Live leaf TP concentrations in *Cladium* were similar to those determined from April 1998 samples. Concentrations in March 2000 ranged between 310 ($N_{1.0C}$) and 538 mg/kg ($S_{4.0C}$). Dead leaf TP concentrations were similar among all years but much lower than live leaf concentrations with all values < 300 mg/kg. Root TP was also low (< 300 mg/kg) and although variable, statistically similar at most stations among years.

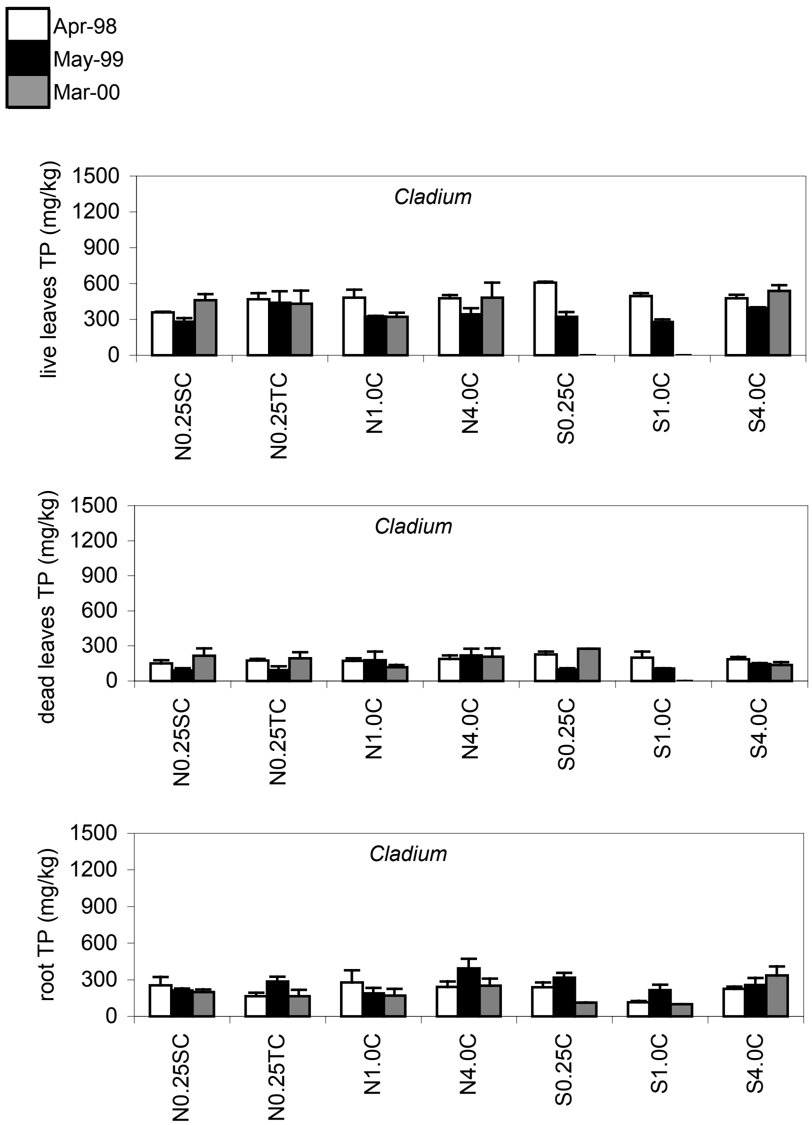


Figure 29. TP concentrations in *Cladium* biomass components within Cladium (C)-designated habitats by station and date.

Typha TP (Figure 30) - Although higher than *Cladium*, nutrient concentrations in *Typha* at this time may have been lower than normal due to a decline in vigor from drought conditions. By station, TP concentrations in *Typha* live leaves were either significantly lower or similar to April 1998 concentrations but consistently lower than

May 1999 values, ranging between 465 ($N_{1.0C}$) and 1,070 mg/kg ($N_{1.0C}$). Dead leaf TP concentrations were low (< 300 mg/kg) compared to live leaves but significantly higher in March 2000 than in May 1999 at stations $N_{4.0T}$ and $S_{4.0T}$. Root TP varied little, ranging between 351 ($N_{0.25C}$, $N_{1.0T}$) and 375 mg/kg ($N_{1.0C}$).

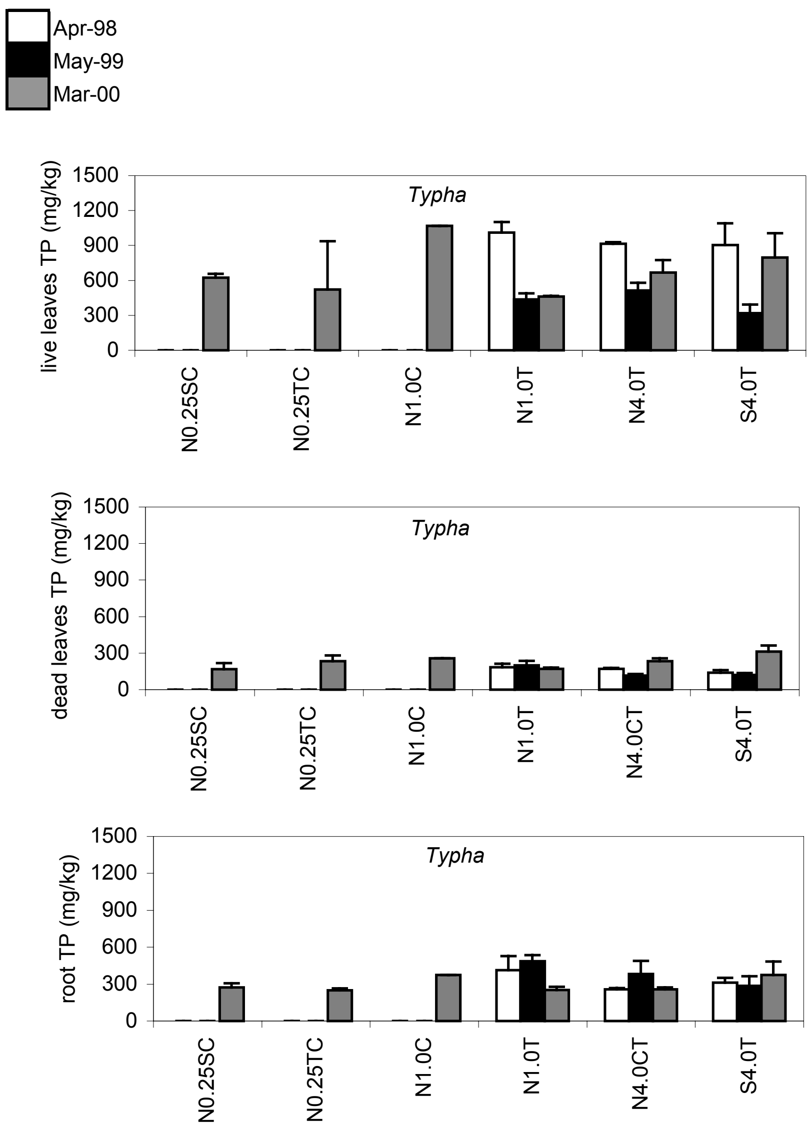


Figure 30. TP concentrations in *Typha* biomass components within *Cladium* (C)- and *Typha* (T)-designated habitats by station and date.

Non-Typha/Cladium TP (Figure 31) - Non-*Typha*/*Cladium* species consisted mostly of *Solidago* spp., *E. capillifolium*, and assorted grasses. TP concentrations in these plants were similar to live leaf concentrations in *Typha* although extremely high concentrations were present at stations $S_{0.25C}$ and $N_{0.25TC}$ in March 2000. Among all stations and years, concentrations ranged between 364 ($S_{4.0T}$, April 1998) and 2,203 mg/kg ($S_{0.25C}$, March 2000).

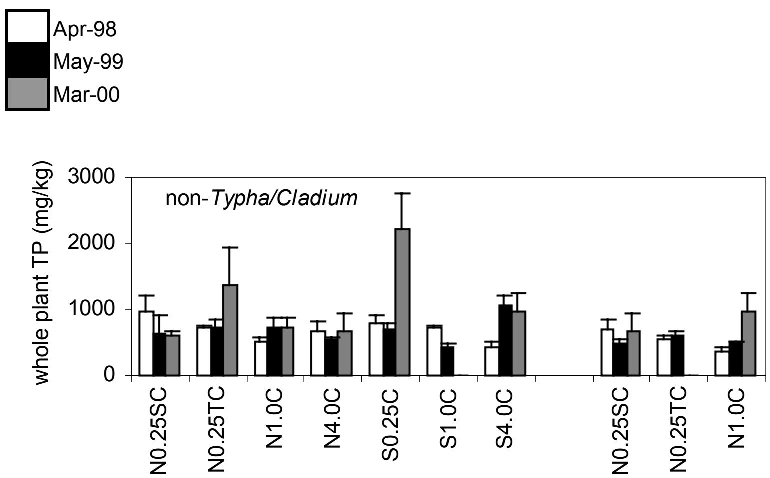


Figure 31. Whole plant TP concentrations in non-*Typha/Cladium* plants within *Cladium* (S)- and *Typha* (C)-designated habitats by station and date.

Cladium TN (Figure 32) - Concentrations of live leaf TN in *Cladium* varied relatively little, ranging between 11,013 ($N_{4.0C}$) and 13,120 mg/kg (N0.$_{25TC}$) and statistically similar to April 1998 and May 1999 values. Dead leaf TN concentrations were approximately twofold lower and fairly similar among years, ranging between 4,697 ($S_{0.25C}$, May 1999) and 9,023 mg/kg ($S_{4.0C}$, May 1999). *Cladium* root TN concentrations in March 2000 were similar to April 1998 values but significantly lower than May 1999 concentrations at many stations. Among all stations and years, root concentrations ranged between 4,787 ($S_{0.25C}$, March 2000) and 10,333 mg/kg ($S_{0.25C}$, March 2000).

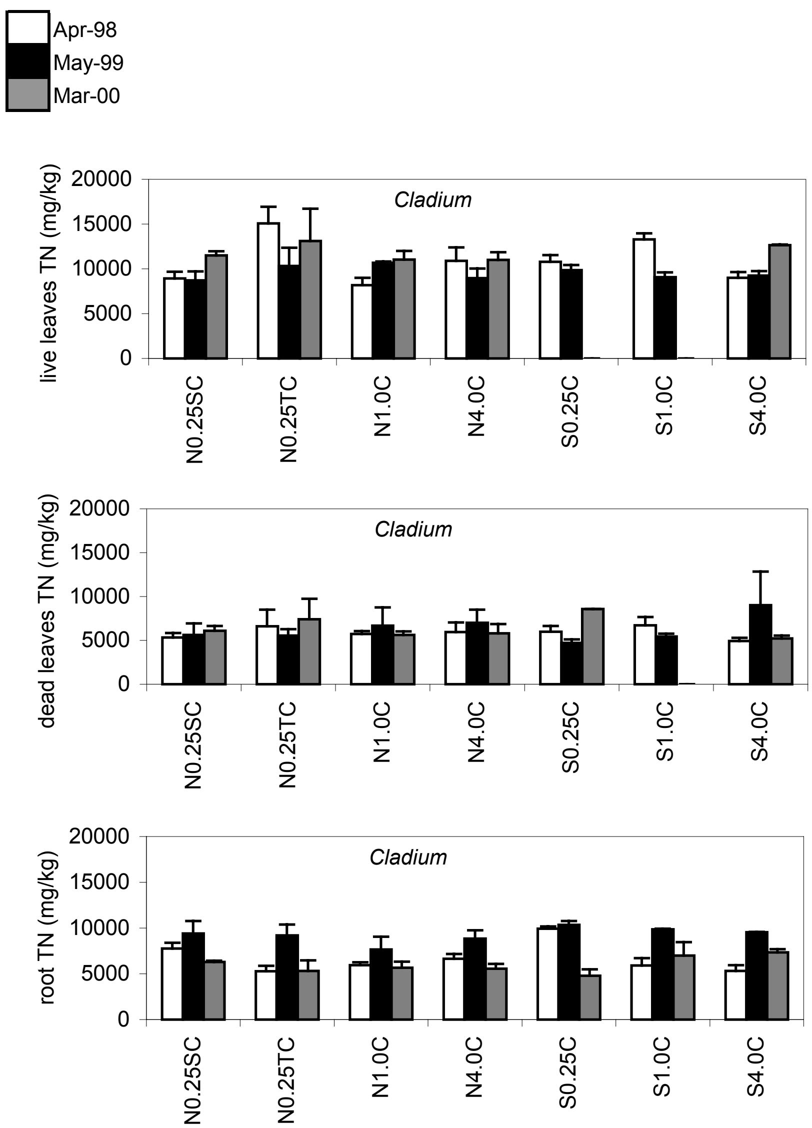


Figure 32. TN concentrations in *Cladium* biomass components within *Cladium* (C)-designated habitats by station and date.

Typha TN (Figure 33) - Live leaf TN in *Typha* ranged between 7,020 ($N_{0.25TC}$) and 11,000 mg/kg ($N_{1.0C}$). Like *Cladium*, dead leaf concentrations were approximately half that of live leaves, ranging between 4,333 ($N_{1.0T}$) and 7,840 mg/kg ($N_{1.0C}$). At $N_{4.0C}$ and $S_{4.0C}$, March 2000 values were significantly higher than April 1998 and May 1999 values. At $N_{1.0C}$, no statistical differences among years were evident. Concentrations of root TN also were much lower than live leaf TN, but similar to dead leaf TN. In this regard, concentrations ranged between 5,235 ($N_{4.0T}$) and 9,205 mg/kg ($N_{0.25SC}$). No consistent patterns of temporal variability were evident for this constituent.

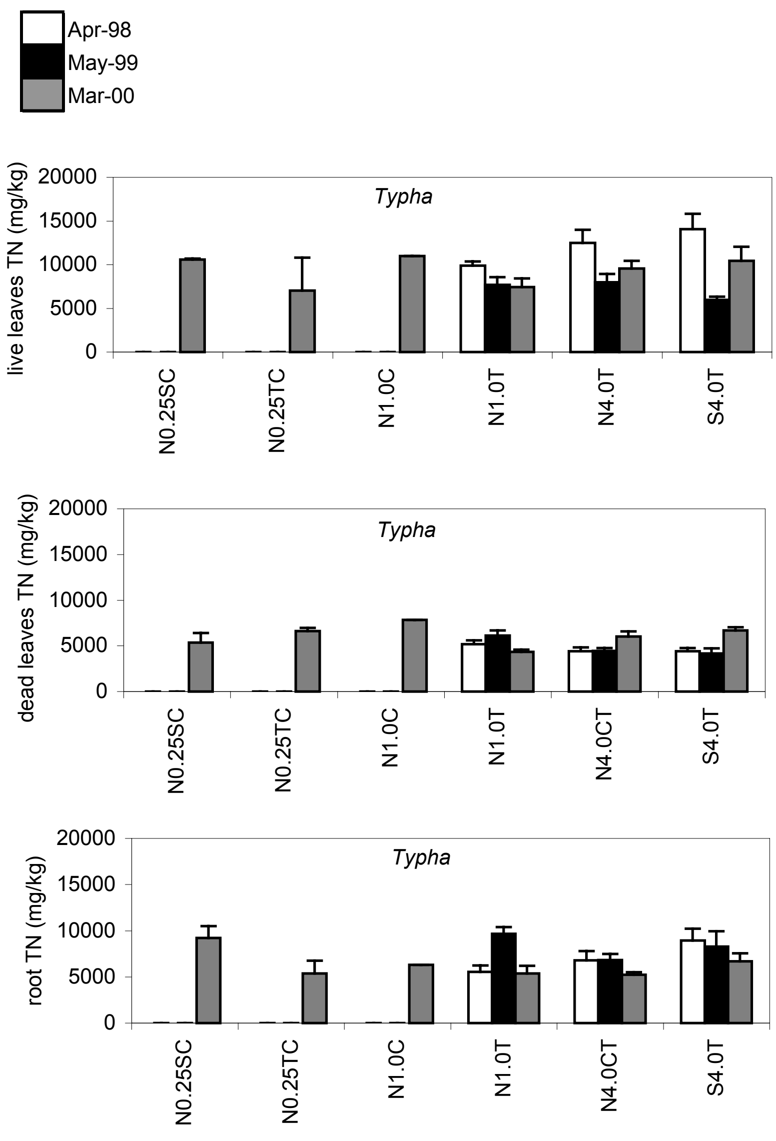


Figure 33. TP concentrations in *Typha* biomass components within *Cladium* (C)- and *Typha* (T)-designated habitats by station and date.

Non-Typha/Cladium TN (Figure 34) - With the exception of $N_{0.25SC}$ (April 1998) and $S_{0.25C}$ (March 2000), TN in non-*Typha/Cladium* plants was quite similar among stations and years. With these station included, however, concentrations ranged between 8,617 ($S_{1.0C}$) and 33,200 mg/kg ($S_{0.25C}$).

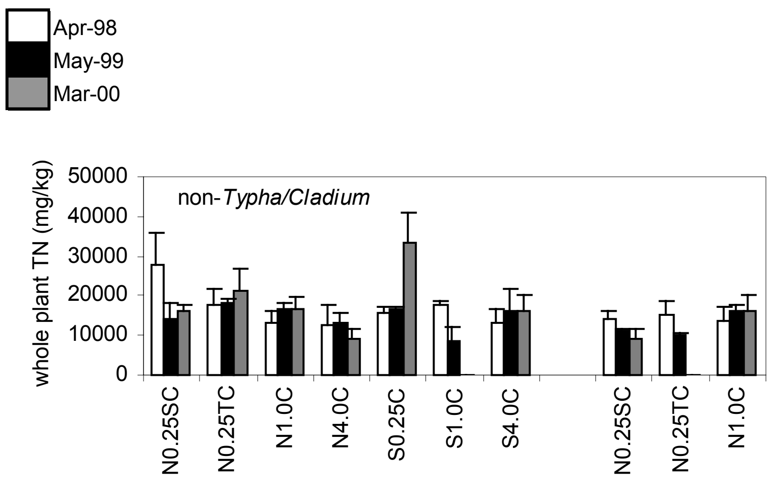


Figure 34. TP concentrations in non-*Typha/Cladium* biomass components within *Cladium* (C)- and *Typha* (T)-designated habitats by station and date.

<u>General species inventory</u>

Since March 1998, species inventories have been compiled for each station and have updated on numerous occasions. The number of species growing within the immediate vicinity of the monitoring stations has varied considerably from 1998 to 2000, primarily as a consequence of the May 1999 fire and 2000 drought. Accordingly, the general macrophyte survey for the RRP was separated into pre- and post-fire periods. It should be noted that some of the 1998 species designations reported in Smith et al. (1998) have since been revised. Additionally, a number of species have lumped together under genera designations (e.g., *Rhynchospora* spp.) where identifications to species could not be made. These designations (listed in Appendix I) were counted as a single species in the following analyses.

In 1998 the total number of species observed at the monitoring stations was 43. This number was reduced to 39 by pre-fire1999 and declined further to 37 by post-fire1999. By 2000, however, the total number of species had increased to 59.

In 1998, individual stations averaged 12 species, ranging between 7 ($N_{2.0}$) and 19 ($S_{4.0}$). In pre-fire 1999, the average was 13, ranging between 7 ($S_{1.0}$) and 19 ($N_{0.5}$) species. There were 11 species per station in post-fire 1999, ranging between 1 ($S_{1.0}$) and 22 ($N_{0.5}$). In the 2000 survey, an average of 23 species per station was recorded, ranging between 19 ($S_{2.0}$) and 30 ($N_{4.0}$).

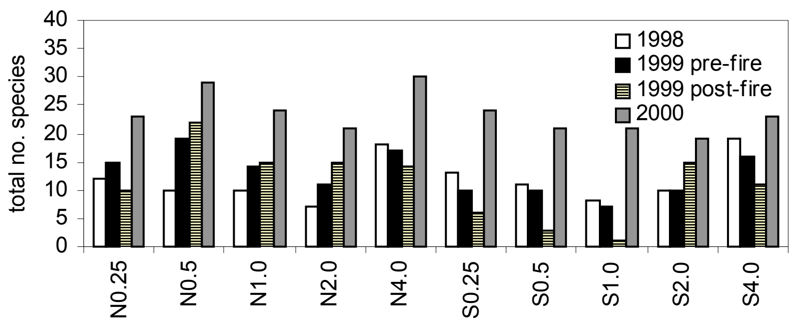


Figure 35. Total number of species observed during general surveys (including genera where identification could not be made to species) by station in 1998, 1999, and 2000.

Percentages of species designated as obligate (OBL), facultative wet (FACW) or facultative (FAC) are shown in Figure 36. %OBL was similar in 1998 (52.8%), pre-fire 1999 (52.8%), and post-fire1999 (52.6%). By 2000, however (after a period of extreme drought), %OBL had decreased to 41.8%.

In 1998 and pre-fire 1999, % FACW values were similar with values of 36.1% and 33.3%, respectively. However, % FACW increased to 42.1% by post-fire 1999 and to 47.3% by 2000. FAC plants increased from 11.1% in 1998 13.9% in pre-fire 1999 and then decreased to 5.3% by post-fire 1999. However, the 2000 surveys indicated another increase to 8.5% as post-fire revegetation continued.

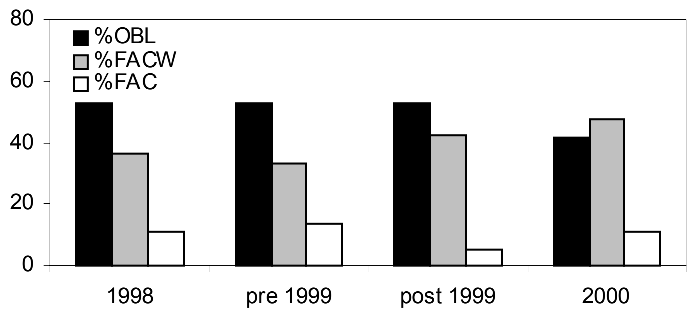


Figure 36. Percentages of species classified as obligate (OBL), facultative wet (FACW) or facultative (FAC) by year.

Species that have not reappeared at the monitoring stations since the May 1999 fire include: *Acer rubrum* (FACW), *Myrica cerifera* (FACW), *Salix caroliniana* (OBL), *Hypericum fasciculatum* (OBL), *Proserpinica paulustris* (OBL), *Rhynchospora tracyi* (OBL), *Rhynchospora haspan* (OBL), *Rorripa terres* (OBL), *Ludwigia octovalis* (OBL). New species that were observed after the fire included: *Cyperus haspan* (OBL), *Cyperus polystachyos* (FACW), *Cyperus surinamensis* (FACW), *Echinochloa walteri* (FACW), *Eustachys glauca* (FACW), *Leptochloa fascicularis* (FACW), *Panicum rigidulum* (FACW), *Iresine rhizomatosa* (FACW), *Setaria geniculutaum* (FACW), *Solanum capsicoides* (non-indicator), *Verbena hastata* (FAC), *Vicia minutiflora* (FAC), *Amaranthus australis* (OBL).

Some other notable changes in macrophytes include the appearance of *T. domingensis*, which did not occur at stations $S_{0.25}$ through $S_{2.0}$ prior to the May 1999 fire but was present at these locations several months later. Furthermore, the dominant species at $S_{0.25}$, $S_{0.5}$, $S_{1.0}$ and $N_{0.25}$ changed from a *Cladium*/*Solidago* spp. mix in May 1998 and March 1999 to *T. domingensis* in November 1999 to virtually monospecific *E. capillifolium* in September 2000 (Figure 37).

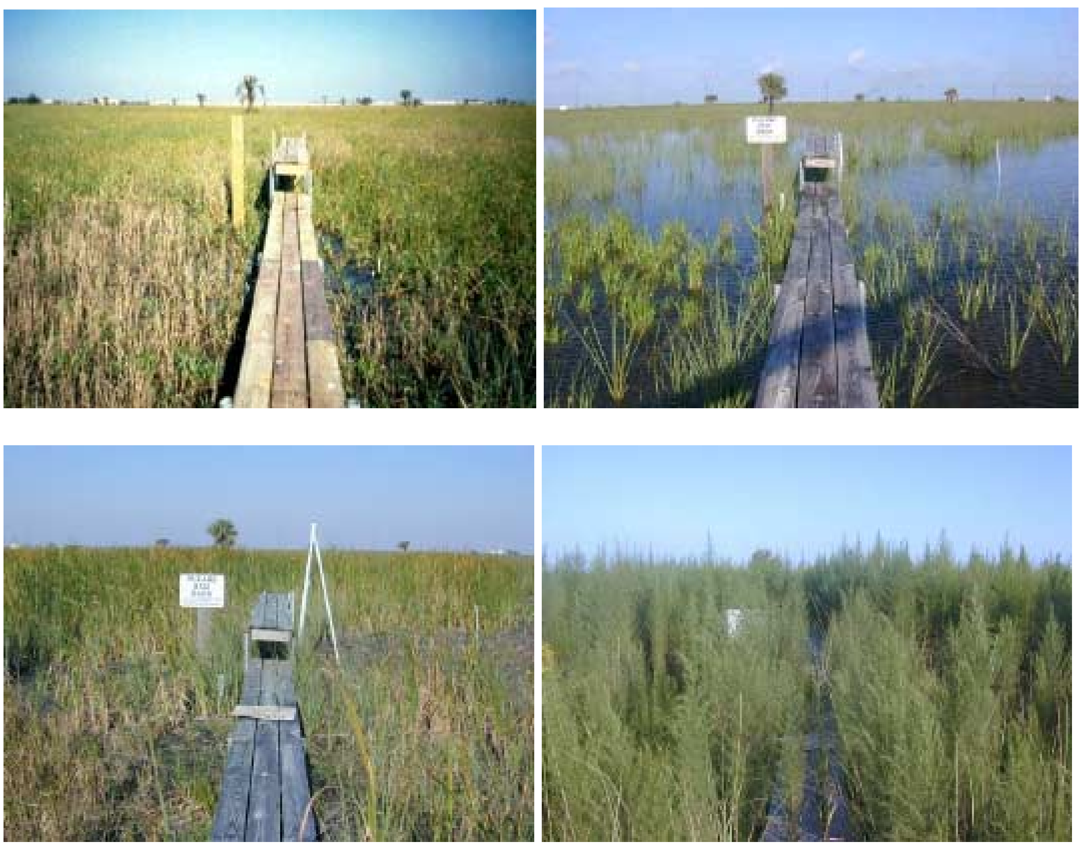


Figure 37. Photographs documenting changes in macrophyte vegetation at station $N_{0.25}$ from *Cladium*/*Solidago* in March 1998 (pre-fire; upper left) and September 1999 (post-fire; upper right), to *Typha* in February 2000 (lower left), and to *Eupatorium* in August 2000 (lower right).

Modeling

Due to the unique hydrological features of the RWMA, there are not appropriate data to calibrate the net settling rate and the relationship between P accretion and soil accretion rates. Furthermore, there are no clear relationships between soil TP concentration and vegetation type. Consequently, the logistic equation in the Everglades Phosphorus Gradient Model could not be recalibrated. As the system evolves in response to STA discharge and more data become available, the EPGM will be recalibrated for parameters specific to the RWMA.

Discussion

Any evaluation of the impacts of rainwater retention is seriously compromised by the confounding effects of fire, drought, and Hurricane Irene. These factors contribute to large-scale variability in the physical, chemical, and biological data. Nevertheless, some basic conclusions can be drawn. Clearly, the RWMA suffers from an insufficient supply of water, which greatly affects the nature of periphyton and plant communities - both directly by water depth and duration and indirectly by creating an environment conducive to fire and soil oxidation/compaction.

In terms of water quality, concentrations of the major ions and metals showed some temporal variation but no consistent trends were evident from the standpoint of increased rainwater retention. For surface water nutrients, species of inorganic N (i.e., NH_4, NO_2, NO_x) exhibited large increases in October following Hurricane Irene, which subsequently declined to pre-hurricane levels by November. Organic N (TKN, TKN-F) showed little temporal variation. Concentrations of PO_4 were below or near detection limits and decreasing trend in TP and TP-F was evident over time. Reductions in N and P species are thought to reflect microbial processes and assimilation by periphyton and macrophyte communities, both of which were undergoing recovery from the May 1999 fire.

In general, periphyton development within the RWMA is limited by macrophyte cover and insufficient surface water. For example, periphyton biomass in October 1998 was relatively low since surface water had only been present for a short time prior to sampling and macrophyte density was high over most of the landscape. By August 1999, however, the presence of surface water combined with increased light availability (from the elimination of macrophytes by fire) allowed periphyton to accumulate, primarily as benthic mats (epipelon). This translated into larger fluctuations in diel DO concentrations reflecting an increased contribution of periphyton metabolism.

Tissue P levels in periphyton were extremely high in August 1999 compared to October 1998. Concentrations also were very high relative to those normally found in periphyton from unimpacted Everglades regions (McCormick, 1996), indicating high P availability. In this regard, periphyton were likely able to exploit P fluxes from the sediment to the overlying water column, particularly in muck-burned areas where large amounts of organic P had been converted to inorganic P (Smith et al. 2000). Taxonomic analyses support this assumption, as community composition was indicative of nutrient enrichment. At the time of this report (October 2000), periphyton is again very scarce in the RWMA due to high levels of macrophyte cover and the prolonged absence of surface water for much of the year.

Macrophyte biomass and species composition is highly variable in the RWMA due to the high frequency of physical disturbance. Drought conditions during 2000 eliminated many obligate wetland plants in favor of species more tolerant of dry soil conditions (FACW, FAC). Fire has played direct (elimination of desirable vegetation) and indirect (creating landscape depressions that are enriched in inorganic nutrients) roles in shaping macrophyte communities (Newman et al., 1998; Sasse et al., 1998). By augmenting water levels using STA-5 discharge, however, it is expected that extreme desiccation and, therefore, fire events will diminish in frequency.

In general, the potential for plants with high growth rates and nutrient requirements to expand within the RWMA is greatly enhanced by antecedent soil conditions (Smith and Newman 2000, attachment B). Given soils that are already enriched, hydrology will determine which species will predominate upon reflooding. Accordingly, *Typha* populations may increase for a period of time until nutrients become more unavailable as buried peat - a process that can only occur under flooded conditions. At some point, this biogeochemical process will theoretically place *Typha* at a competitive disadvantage relative to other plant species that are adapted to more oligotrophic conditions.

There are, in addition to creating appropriate hydrologic conditions for sequestering nutrients in peat, other potential benefits of STA-5 discharges. These include the encouragement of other non-*Typha* wetland vegetation (e.g., *Sagittaria*, *Utricularia*, *Pontederia*, etc.) and the provision of living space for aquatic invertebrates, fish, reptile, and amphibians. Increasing the abundance of these lower trophic level organisms may then translate into healthier avian and mammal populations. In conclusion, although hydrologic restoration of the RWMA may produce some undesirable short-term outcomes (such as *Typha* proliferation), the establishment of a longer hydroperiod using

STA-5 discharges is a critical component to meeting the long-term restoration goals for this region and for the Everglades as a larger, integrated system.

Literature cited

Browder, J.A., P.J. Gleason, and D.R. Swift. 1994. Periphyton in the Everglades: spatial variation, environmental correlates, and ecological implications, pp. 379-418. In: S.M. Davis and J.C. Ogden (eds), Everglades: the Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, Florida.

Grimshaw, H.J., L.J. Wenkert, M. Brandenburg, K. Segerblom, G.A. Marsh, and W. Charnetzky. 1993. Effect of shading by emergent macrophytes on the productivity of natural Everglades periphyton communities. Archiv fur Hydrobiologie 128: 257-276.

McCormick, P.V. and M. O'Dell. 1996. Quantifying periphyton responses to phosphorus in the Florida Everglades: a synoptic-experimental approach. Journal of the North American Benthological Society 15(4): 450-468.

McCormick, P.V. 1996. Periphyton-water quality relationships along a nutrient gradient in the northern Florida Everglades. Journal of the North American Benthological Society 1(4): 433-449.

Miao, S.L. and F.H. Sklar. 1998. Biomass and nutrient allocation of sawgrass and cattail along a nutrient gradient in the Florida Everglades. Wetland Ecology and Management 5: 245-263.

Smith, S.M., P.B. Garrett, J.A. Leeds, and P.V. McCormick. 1999. Baseline water quality, soil chemistry, periphyton, and macrophyte data from the Rotenberger Wildlife Management Area, 1998-1999. Report submitted to the U.S. Army Corps of Engineers. South Florida Water Management District, West Palm Beach, Florida.

Swift, D.R. and R.B. Nicholas. 1987. Periphyton and water quality relationships in the Everglades Water Conservation Areas, 1978-1982. Tech. Publ. 87-2.

Walker, William W., Jr. and Robert H. Kadlec. 1996. A Model for Simulating Phosphorus Concentrations in Waters and Soils Downstream of Everglades Stormwater Treatment Areas. Prepared for U.S. Department of the Interior, August 13, 1996.

Wood, E.J.F. and N.G. Maynard. 1974. Ecology of the micro-algae of the Florida Everglades, pp. 123-145. In: P.J. Gleason (ed.) Environments of South Florida: Past and Present, Memoir No. 2. Miami Geological Society, Coral Gables, FL.

Appendix I. List of macrophyte species observed (all stations) in the 1998, pre-fire 1999, post-fire 1999, and 2000 general surveys.

1998	1999 pre-fire	1999 post-fire	2000
Andropogon glomeratus	Acer rubrum	Andropogon glomeratus	Amaranthus australis
Baccharis glomerulifolia	Andropogon glomerulatus	Boehmeria cylindrica	Andropogon glomerulatus
Boehmeria cylindrica	Aster elliottii	Centella asiatica	Baccharis glomerulifolia
Cirscium nuttallii	Baccharis glomerulifolia	Cephalanthus occidentalis	Boehmeria cylindrica
Cladium jamaicense	Boehmeria cylindrica	Cladium jamaicense	Centella asiatica
Conoclinium coelestinum	Centella asiatica	Cyperus haspan	Cephalanthus occidentalus
Cyperus spp.	Cladium jamaicense	Cyperus polystachyos	Cladium jamaicense
Dichromena colorata	Cyperus spp.	Cyperus surinamensis	Conoclinium coelestinum
Erianthus giganteus	Dichromena colorata	Dichromena colorata	Cyperus spp
Erigeron quercifolious	Erianthus giganteus	Eleocharis (baldwinii)	Cyperus haspan
Eupatorium capillifolium	Erigeron quercifolia	Eleocharis cellulosa	Cyperus odorata
Euthamia minor	Eupatorium capillifolium	Erianthus giganteus	Cyperus polystachyos
Hydrocotyl bonariensis	Euthamia minor	Euthamia minor	Dicromena colorata
Hydrolea corymbosa	Hypericum fasciculatum	Hydrocotyl bonariensis	Diodia virginiana
Hypericum spp.	Ludwigia (alata?)	Ipomea sagittata	Echinochloa walteri
Ipomea sagittata	Ludwigia repens	Leptochloa fascicularis	Eleocharis (baldwinii?)
Ludwigia octovalvis	Lythrum alatum	Ludwigia (alata?)	Eleocharis cellulosa
Ludwigia peruviana	Mikania scandens	Ludwigia (microcarpa?)	Erianthus giganteus
Ludwigia repens	Mitreola petiolata	Ludwigia peruviana	Erigeron querqifolius
Ludwigia spp.	Myrica cerifera	Ludwigia repens	Eupatorium capillifolium
Lythrum alatum	Oxalis spp.	Lythrum alatum	Eupatorium mikanoides
Mikania scandens	Panicum dichotomon	Mikania scandens	Eustachys glauca
Myrica cerifera	Panicum hemitomon	Panicum hemitonon	Euthamia minor
Oxalis spp.	Peltandra virginica	Panicum rigidulum	Hydrocotyle bonariensis
Panicum dichotomum	Phyla nodiflora	Paspalidium geminatum	Hydrolea corymbosa
Panicum hemitomon	Pluchea rosea	Paspalum spp.	Hyptis alata
Paspalidium geminatum	Polygonum hydropiperoides	Peltandra virginica	Ipomea sagittata
Panicum spp.	Proserpinaca palustris	Pluchea rosea	Iresine rhizomatosa
Peltandra virginica	Rhynchospora microcarpa	Rhynchospora inundata	Leptochloa fasicularis
Pluchea rosea	Rorripa teres	Rhynchospora microcarpa	Ludwigia alata
Polygonum spp.	Sagittaria lancifolia	Sagittaria graminea	Ludwigia microcarpa
Rhynchospora (tracyi?)	Salix caroliniana	Sarcostemma clausum	Ludwigia (peruviana?)
Rhynchospora haspan	Sarcostemma clausum	Solidago leanvenworthii	Ludwigia repens
Rhynchospora inundata	Senecio glabellus	Solidago stricta	Lythrum alatum
Rhynchospora microcarpa	Solidago leavenworthii	Teucrium canadense	Mikania scandens
Rorripa teres	Solidago stricta	Thelypteris palustris	Mitreola petiolata
Sagittaria lancifolia	Teucrium canadense	Typha domingensis	Oxalis spp
Salix caroliniana	Thelypteris palustris		Panicum dichotomum
Solidago leavenworthii	Typha domingensis		Panicum hemitomon
Solidago stricta			Panicum rigidulum
Teucrium canadense			Paspalidium geminatum
Thelypteris palustris			Peltandra virginica
Typha domingensis			Phyla nodiflora
			Pluchea odorata
			Pluchea rosea
			Polygonum hydropiperoides
			Rhynchospora microcarpa

2000 (continued)

Sagittaria lancifolia
Sarcostemma clausum
Senecio glabellus
Setaria geniculata
Solanum capsicoides
Solidago leavenworthii
Solidago stricta
Teucrium canadense
Thelypterus paulustris
Typha domingensis
Verbena hastata?
Vicia minutiflora

Attachment A.

Effects of above- and below-ground fire on soils of a northern Everglades marsh

*Stephen M. Smith, Susan Newman, Patrick B. Garrett, and Jennifer A. Leeds

* Correspondence: Everglades Division, Watershed Research and Planning Department, South Florida Water Management District, 3301 Gun Club Road, West Palm Beach, Florida 33406. Tel: (561) 682-2844, email: stephen.smith@sfwmd.gov

Target journal: Biogeochemistry

Keywords: fire, soils, surface-burn, muck-burn, nutrients, phosphorus, cattail

Abstract

The effects of above- (surface) and below-ground (muck) fire on a number of soil constituents were examined within a hydrologically-altered marsh in the northern Florida Everglades. Muck fire resulted in losses of total carbon, nitrogen, and organic forms of phosphorus while inorganic phosphorus was elevated. In addition, muck fire resulted in increased vertical heterogeneity in concentrations of most constituents between upper and lower sediment layers. Surface fire had a limited impact on the measured constituents. The effect of physical versus chemical processes during burning were assessed using ratios of constituent:total calcium concentrations. In this context, increases in the levels of inorganic P fractions in muck-burned areas were due to the physical reduction of soil while decreases in N and C were the result of volatilization. In an ecological context, the observed soil transformations may encourage the growth of opportunistic plant species such as <u>Typha</u> <u>domingensis</u> (cattail).

Introduction

Fire plays a critical role in both the preservation and evolution of wetland ecosystems (Wade 1980; Kushland 1990; Cohen 1994). However, studies on the effects of fires in wetlands are relatively few compared to other habitats. Moreover, research has largely focussed on the recovery of flora and fauna while information on physical parameters of the environment that may be affected and subsequently influence these processes is lacking (Vogl 1973; van Arman & Goodrick 1979; Abrahamson 1984; Smith & Kadlec 1985; Lee et al. 1995).

By definition, wetlands have standing water for extended periods of time and fires often consume only a portion of above-ground vegetation while roots and peat layers remain protected underwater (i.e., surface burn). Upon severe drydown, however, the combustion of soil organic matter can occur (muck burn). The obvious consequences of muck burns include the elimination of existing vegetation through root damage, destruction of the natural seed bank, and alteration of hydroperiod by lowering soil elevations. Lesser-known effects may involve changes in the pool-size and/or availability of various soil constituents.

In general, fires tend to alter the availability of nutrients that are otherwise stored in above- and belowground plant matter (Faulkner & de la Cruz 1982; Vogl 1969; Wilbur & Christensen 1983). In some cases, levels of soil nutrients increase after a fire (Marion et al. 1991; Tomkins et al. 1991; Kutiel & Shaviv 1993). Alternatively, no enrichment may be observed (Laubhan 1995; Weiss 1980) or nutrients may be lost through volatilization and/or export of wind-dispersed ash (Kutiel & Shaviv 1993; Marion et al. 1991; Turner et al. 1997). The extent of soil modification is largely dependent upon fire temperature, which itself is influenced by fuel moisture levels (Robichaud & Waldrop 1994), fuel loads in the soil (Scott & Van Wyk 1990), and plant species (Scott & Van Wyk 1990; Kutiel & Shaviv 1991).

In the Florida Everglades, fire has been essential to the maintenance of sawgrass (Cladium jamaicense Crantz.) prairies, the dominant habitat within the marsh, mainly through the periodic elimination of successional species (Loveless 1959). However, the frequency and nature of fire events in the Everglades is believed to have been altered by water management practices over the last several decades (Robertson 1953; Alexander & Crook 1974; Gunderson and Snyder 1994). The change in fire patterns has been discussed as one possible mechanism by which opportunistic plant species have proliferated (Christensen & Burrows 1986). In particular, it has been hypothesized that replacement of sawgrass vegetation by invading cattail (Typha domingensis Pers.) has been encouraged by occurrences of muck fires in the Everglades (Davis 1943; Newman et al. 1998). To what extent fire-induced changes in soil properties has contributed to this pattern of replacement is unclear.

Aside from prescribed burns, it difficult to predict when and where a fire will occur. Accordingly, the interpretation of fire effects is often constrained by a lack of pre-fire baseline data. Additionally, comparisons of pre- and post-fire data that have been collected from different sampling locations can be confounded by spatial variability in soil properties. In the summer of 1999, we examined the effects of both muck and surface fire on soils of a northern Everglades marsh from which baseline data had been previously obtained. Our main objective was to determine the extent of soil transformation (particularly nutrients) by each type of fire. We hypothesized that P availability would increase in both surface- and muck-fire through mineralization, whereas C and N content would likely decrease in muck-fire through volatilization. One difficulty, however, in making nutrient chemistry comparisons is that soil profiles at burned sites are confounded by the presence of plant ash, and probably to a greater extent, by the physical concentration of soil layers into smaller volumes (observed as soil loss) during muck fire. Without an accurate measure of the amount (depth) of soil lost, it is impossible to directly assess the quantity of nutrients volatilized or to account for any concentration increase caused by the consolidation of remnant soil (ash) into a smaller volume. Therefore, we compared nutrients of interest with calcium since this element 1) did not have a downcore gradient within the top 0-20 cm of soil (Smith et al. 1999 RTB report) and 2) has melting and boiling points of 850 and 1440°C, respectively, and so requires extremely high temperatures to be lost through volatilization (Merck 1989). Thus, calcium concentrations in the upper layer would increase as it is retained within a smaller volume of soil. Accordingly, constituent:total calcium (TCa) ratios provided a relative measure of how changes were related to physical (i.e., concentrating mechanism of soil loss) and chemical (i.e., volatilization) processes.

Methods

Study area - The Rotenberger Wildlife Management Area (RWMA) encompasses approximately 28,000 acres of land in the northwest Everglades (Figure 1). The area is fed solely by rainfall and has, for decades, been overdrained. For example, Newman et al. (1998) calculated that only about 1% of the RWMA has standing water for > 9 months per year. These conditions have resulted in elevated concentrations of nutrients per unit volume of soil as a result of compaction by desiccation, oxidation, and mineralization by fire (Newman et al. 1998). Although the RWMA once was dominated by sawgrass (Davis 1943), cattail has become extremely abundant, and its current distribution is associated with areas of previous muck burns (Newman et al. 1998; Sasse 1999). On May 23, 1999, a lightning strike ignited a large fire that swept through the marsh. Surface and muck burns were evident at a number of previously established sampling sites that were relocated by remnant sampling markers.

Sampling - In February 1998, soil was collected from 10 different sites in the RWMA (Figure 1) as part of a separate monitoring program implemented to document the process of hydropattern restoration. At each site, sampling was conducted within stands of sawgrass, cattail, or mixed-grass assemblages. On June 2, 1999, after the tract had burned, soil samples were collected from the same locations. No standing water was present at the time of pre- or post-fire sampling. Triplicate soil samples were collected using a 10-cm diameter aluminum coring tube. The tube was placed on the soil surface and a serrated knife was used to cut around its circumference as a means to avoid soil compaction during core insertion. Cores were extruded and sectioned into 0-2 and 2-10 cm depth layers.

Samples were sealed in plastic bags, placed on ice, and stored at 4°C until they could be analyzed for total carbon (TC), total nitrogen (TN), TCa, total phosphorus (TP), percent ash content (% Ash), and bulk density (BD). Additionally, through a series of sequential extractions, concentrations of specific phosphorus fractions were analyzed. These included labile inorganic phosphorus (KCl-SRP), calcium- and magnesium-bound phosphorus (HCl-SRP), iron- and aluminum-bound phosphorus (NaOH-SRP), and total alkali-extractable phosphorus (NaOH-TP). The organic fraction of NaOH extractable P is the difference between NaOH-SRP and NaOH-TP (NaOH-P_o), although some loss of Po is likely to occur with each sequential extraction. Total inorganic P (TP_i) was calculated as the sum of KCl-SRP, HCl-SRP, and NaOH-SRP. Analyses were conducted according to the procedures of Page et al. (1982), USEPA (1983), USACOE (1986), and Reddy et al. (1991).

Data analyses - Soil constituents varied little among vegetation types prior to burning. Accordingly, the data (both pre- and post-fire) were grouped according to whether they were sampled from locations that were subsequently surface-burned (SB, n=27) or muck-burned (MB, n=34). As such, the complete set of samples was comprised of soils from the following categories: 1) pre-fire, surface-burned area (PRE_{SB}), 2) post-fire, surface-burned area ($POST_{SB}$), 3) pre-fire, muck-burned area (PRE_{MB}), and 4) post-fire, muck-burned area ($POST_{MB}$). The number of post-fire samples analyzed for specific P fractions was reduced by one third due to analysis costs. All data were corrected for bulk density and are expressed on a volume basis ($\mu g/cm^3$). Constituent:TCa ratios were also calculated. For statistical analysis, the data were transformed logarithmically to improve normality and heteroscedasticity and subjected to analysis of variance (split-plot design). Specific comparisons between pre- and post-fire muck- and surface-burned soils were examined using statistical contrasts (SAS 1989).

Results

0-2 cm layer - Pre-fire bulk density was higher in the muck- vs. surface-burned area. However, none of the measured soil constituents in this layer showed any significant pre-fire spatial (between burn areas) variability. Increasing trends in BD and % ash were observed in response to both types of fires, but were only significant for MB areas (Figures 2a,b). TN and TC both exhibited a slight increase after surface burning and a decrease following muck burning (Figures 2c,d), whereas TCa (Figure 2e) exhibited more than a four-fold increase in response to muck-burning but relatively little change after surface burning. This response illustrates of the concentrating mechanism of muck-burning as the soil collapses into a smaller volume.

TP concentrations increased slightly in SB areas and significantly in MB areas, resulting in $POST_{MB}$ soils having higher concentrations than $POST_{SB}$ soils (Figure 3a). KCl-SRP concentrations increased substantially in response to both fire types (Figure 3b). HCl-SRP exhibited a slight increase in response to surface-burning and a significant increase in response to muck-burning (Figures 3c). In contrast, NaOH-SRP decreased in $POST_{SB}$ soils and no change was observed in $POST_{MB}$ soils (Figure 3d). Post-fire NaOH-P_o concentrations within both SB and MB areas were greatly lowered (Figure 3e).

2-10 cm layer - Pre-fire concentrations of most 2-10 cm layer soil constituents were lower than in the 0-2 cm layer in areas that were subsequently muck-burned. In surface-burned areas, concentrations of TCa, TN, and TC were similar between the two layers while all forms of P exhibited a significant reduction with depth (Figures 2 and 3). Only TP and NaOH-SRP showed significantly differences in pre-fire soils, with PRE_{SB} concentrations being higher than PRE_{MB}.

On a spatial basis, the BD of the 2-10 cm layer was similar between PRE_{SB} and PRE_{MB} soils. However, PRE_{MB} soils were lower than PRE_{SB} soils with respect to TP. Concentrations of TC, TN, TCa, and all P fraction showed no significant pre-fire differences between burn areas.

Bulk density and % ash increased in muck- but not surface-burned soils (Figures 2f, g). TC and TN were much lower in $POST_{MB}$ vs. PRE_{MB} soils but similar between $POST_{SB}$ and PRE_{SB} soils (Figures 2h,i). Significant fire-related responses of TCa were not observed in this layer (Figure 2j). TP concentrations were slightly reduced in $POST_{SB}$ and $POST_{MB}$ soils compared to the pre-fire values (Figure 3f). A large post-fire increase in KCl-SRP occurred in the muck- but not surface-burned area. A smaller, but significant increase in HCl-SRP also was observed in this regard (Figures 3g,h) while significant reductions were evident for NaOH-SRP and NaOH-P_o in both areas (Figures 3i,j).

Constituent:Ca ratios (0-2 cm layer) - Soil TN:TCa was only slightly reduced from 0.86 to 0.80 during surface fire (Table 1). Soil TC:TCa responded similarly with only a small decrease from 11.94 to 10.71. In contrast, large reductions occurred during muck fire. For example, TN:TCa and TC:TCa were 1.13 and 15.28 in PRE_{MB} soils compared to 0.31 and 3.55 in $POST_{MB}$ soils, respectively. TP:Ca ratios showed little response to either surface or muck fire. TP:Ca was 0.022 in PRE_{SB} soils and 0.018 in $POST_{SB}$ soils. These ratios were 0.018 and 0.014 in PRE_{MB} and $POST_{MB}$ soils. However, NaOH-P_o:TCa decreased considerably in muck burned soils, with ratios of 15.10 and 1.36 in PRE- vs. $POST_{MB}$ soils. In surface-burned soils, this reduction was less pronounced as values decreased from 11.92 to 6.37. HCl-SRP:TCa exhibited a large increase from 5.97 to

18.50 in response to muck burning but changed little after surface burning. KCl-SRP:TCa increased from 0.03 to 0.18 in surface-burned soils but remained virtually unchanged after muck fire.

Inorganic and organic P pools - Figure 4 shows the relative proportions of the different P fractions for the 0-2 cm layer of each soil type. As summarized above, increases in inorganic P corresponded with decreases in organic P. HCl-SRP made up the largest portion of Pi in both surface and muck burned soils. However, although the transformation of P_o to P_i occurred in both types of fire, it was much more extensive in muck-burned soils.

DISCUSSION

The burn pattern of this fire showed some correlation with vegetation type. Cattail stands generally experienced surface burns (3 out of 4 stations), whereas mixed (non-cattail/sawgrass) stands tended to be muck-burned (7 out of 8 stations). Sawgrass stands showed evidence of both surface (5 out of 9 stations) and muck (4 out of 9 stations) burns. This pattern is likely a consequence of elevation differences as cattails in the RWMA occupy areas of lower elevations (commonly regions of previous muck burns) that have longer hydroperiods and hold more soil moisture (Smith et al., 1999). Additionally, cattail leaf tissue may not carry fire as well as other plants due to its relatively high moisture content (S.L. Miao, personal communication).

In the upper (0-2 cm) soil layer, surface fire resulted in little change for most soil constituents with the exception of KCL-SRP and NaOH-SRP, which increased and decreased, respectively. The effects of surface-fire may be more pronounced in oligotrophic regions where small changes in the levels of various constituents would be proportionally large in comparison to low pre-existing concentrations. RWMA soils are already enriched on a volumetric basis due to soil compaction, oxidation, and mineralization. Thus, nutrient additions from above-ground biomass would contribute relatively less to existing pools.

Muck burns had a large effect on soils, most notably in the upper layer. Interpretation of the data is somewhat complicated by the variable influence of physical vs. chemical processes. Absolute comparisons of the same soil layers in pre- vs. post muck-burned soils are impossible due to soil loss and consolidation. Accordingly, post-fire concentration changes in 2-10 cm soil layers of muck-burned areas may simply reflect preexisting downcore concentration gradients. In an ecological sense, however, the mechanism is irrelevant since the transformed soil profile becomes the new habitat structure for revegetation.

Soil TN:TCa and TC:TCa ratios in the 0-2 cm layer did not change appreciably during surface burns, but decreased considerably during muck fire, indicating that both N and C were volatilized in the process. In contrast, the ratio of TP:Ca was very similar in pre- and post-fire soils, indicating that P was not volatilized during the fire. However, the form of P was affected. In muck burned soils, NaOH-P_o:TCa decreased while HCl-SRP:TCa increased; the burning of soil during the muck fire resulted in the conversion of organic P to inorganic P as has been observed in habitats affected by fire (Saa et al. 1993). In surface-burned soils, there was a smaller decrease in the NaOH-Po:TCa ratio and an increase KCl-SRP:TCa. However, total Pi:TCa remained the same given that

KCl-SRP was such a small component of the total Pi pool. Interestingly, the mass of TCa (in grams) in the 0-2 layer of post-fire muck-burned samples divided by the amount in the 0-10 cm (0-2 cm + 2-10 cm) increment of pre-fire muck-burned samples produced a mean value of 1.05, indicating that on average there was just slightly more TCa in the 0-2 layer after muck fire as there was in the 0-10 cm layer before the fire.

Due to the concentrating effect of the muck-burning process, differences between upper and lower layer concentrations of most constituents rose considerably after muck fire. In surface-burned areas, much smaller differences between layers were present with the exception of KCl-SRP. Vertical heterogeneity of this nature, in the absence of an external nutrient loading source (e.g., high-nutrient surface water inflows), may be useful in characterizing the fire history of a particular site. In the pre-fire data, the magnitude of difference between upper and lower layer concentrations were higher in MB than SB areas. A similar pattern existed for bulk density values. This suggests that the PRE_{MB} areas may have been muck-burned in the past.

Ecological implications

Pre-fire soil constituents varied little by vegetation type, probably because over time, microbial, algal, and plant populations can 1) transform soil constituents (i.e., conversion of P_i to P_o), 2) remove constituents (i.e., denitrification), and 3) add constituents (i.e., N_2 fixation) to the system. These processes tend to minimize differences in soil properties that may exist in the early successional stages of a disturbed area. However, subsequent disturbance may effectively re-establish spatial variability that ultimately influences species distributions. For example, an increase in the amount of readily-available P would theoretically permit higher rates of P uptake, translating into enhanced growth of those species with high P requirements such as T. domingensis.

Where existing seed banks have been destroyed, revegetation will presumably be influenced by seed germination responses to upper layer soil chemistry. In this fire, a mosaic of muck and surface burns occurred. Whether increased horizontal variability in soil constituents influences vegetative landscapes is the subject of future study. Under laboratory conditions, however, Rivard & Woodard (1988) reported that the nutrients in T. latifolia ash stimulated seed germination. Typha spp. distributions in the field have also been positively correlated with Ca concentrations (Volk et al. 1975; Adriano et al. 1980). In this study, Ca became concentrated in the upper layers of muck-burned soils. Thus, muck fire in the Everglades may create an ideal environment for T. domingensis with respect to soil elements. The additional influence of altered hydrology and removal of plant competition following severe fire events may further optimize conditions for establishment and growth in this ecosystem.

Literature cited

Abrahamson W (1984) Species responses to fire on the Florida Lake Wales ridge. Amer. J. Bot. 71(1): 35-43.

Adriano DC, Fulenwider A, Sharitz RR, Ciravolo TG, & Hoyt GD (1980) Growth and mineral nutrition of cattail (Typha) as influenced by thermal alteration. J. Environ. Qual. 9(4): 649-653.

Alexander TR & Crook AG (1974) Recent vegetational changes in southern Florida. In: Gleason PJ (Ed) Environments of South Florida: present and past, memoir 2 (pp 61-71). Miami Geological Society, Coral Gables, Florida.

Christensen PE & Burrows ND (1986) Fire: an old tool with a new use. In: Grooves RH & Burdon JJ (Eds) Ecology of biological invasions: an Australian perspective (pp 57-66). Australian Academy of Science, Canberra, Australia.

Cohen AD (1994) Evidence of fires in the ancient Everglades and coastal swamps of South Florida. In: Gleason PJ (Ed) Environments of South Florida: present and past, memoir 2 (pp. 213 to 218). Miami Geological Society, Coral Gables, Florida.

Davis JH (1943) The natural features of southern Florida, especially the vegetation and the Everglades. Florida Geological Survey Bulletin 25. 311 pp.

Faulkner SP & de la Cruz AA (1982) Nutrient mobilization following winter fires in an irregularly flooded marsh. J. Environ. Qual. 11: 129-133.

Gunderson LH and Snyder JR (1994) Fire patterns in the southern Everglades. In: DAVIS SM & OGDEN JC (Eds.) Everglades, the Ecosystem and its Restoration (pp. 307-322). St. Lucie Press, Delray Beach, FL.

Kushland JA (1990) Freshwater marshes. In: Myers RL & Ewel JJ (Eds.) Ecosystems in Florida (pp. 324-362). University of Central Florida Press, Orlando.

Kutiel P & Shaviv A (1989) Effect of simulated forest fire on the availability of N and P in Mediterranean soils. Plant Soil 120: 57-63.

Kutiel P & Shaviv A (1993) Effects of soil type, plant composition, and leaching on soil nutrients following a simulated forest fire. Forest Ecol. Man. 53: 329-343.

Laubhan MK (1995) Effects of prescribed fire on moist-soil vegetation and soil macronutrients. Wetlands 15(2): 159-166.

Lee MA, Ponzio K, & Ormiston BG (1995) Fire effects and fire management in the upper St. Johns river basin marsh, Florida. In: Cerulean SI & Engstrom RT (Eds) Proceedings 19th Tall Timbers Fire Ecology Conference. Fire in Wetlands: A Management Perspective (pp 142-150). Tall Timbers Research, Inc., Tallahassee, Florida.

Loveless CM (1959) A study of vegetation in the Florida Everglades. Ecology 40: 1-9.

Marion GM, Moreno JM, & Oechel WC (1991) Fire severity, ash deposition, and clipping effects on soil nutrients in chaparral: Soil Sci. Soc. Am. J. 55: 235-240.

Merck & CO., Inc. (1989) The Merck index. Rahway, New Jersey, USA.

Newman S, Schuette J, Grace JB, Rutchey K, Fontaine T, Reddy KR & Pietrucha M (1998) Factors influencing cattail abundance in the northern Everglades. Aquat. Bot. 60: 265-280.

Page AL, Miller RH & Keeney DR (1982) Methods of Soil Analysis (Part 2, Chemical and Microbiological Properties). 2nd edition. American Society of Agronomy, Inc. and Soil Science Society of America, Inc. Madison.

Reddy KR, DeBusk WF, Wang Y, DeLaune RD & Koch MS (1991) Physico-chemical properties of soils in Water Conservation Area 2 of the Everglades. Final report to the South Florida Water Management District, West Palm Beach, Florida.

Rivard & Woodward (1988) Light, ash, and pH effects on the germination and seedling growth of Typha latifolia (cattail). Can. J. Bot. 67: 2783-2787.

Robertson WB (1953) A survey of the effects of fire in Everglades National Park. Mimeo. reps. National Park Service. 169 p.

Robichaud PR & Waldrop TA (1994) A comparison of surface runoff and sediment yields from low- and high-severity site preparation burns. Water Res. Bull. 30: 27-34.

Saa A, Trasar-Cepeda MC, Gil-Sotres F & Carballas T (1993) Changes in soil phosphorus and acid phosphatase activity immediately following forest fires. Soil Biol. Biochem. 25(9): 1223-1230.

SAS Institute Inc (1989) SAS/STAT® User's guide, ver. 6 (4th edition). Gary, North Carolina.

Sasse B (1999) Cattail monitoring in the Rotenberger Wildlife Management Area, 1992-1998. Florida game and Freshwater Fish Commission, Wellington, Florida.

Scott DF & Van Wyk DB (1990) The effects of wildfire on soil wettability and hydrological behavior of an afforested catchment. J. Hydrol. 121: 239-256.

Smith LM & Kadlec JA (1985) Comparisons of prescribed burning and cutting of Utah marsh plants. Great Basin Nat. 45(3): 462-467.

Smith SM, Chen Z, Leeds JA, Garrett PB, Brewer JA, McCormick PV & Fontaine T (1999) Baseline water quality, soil, periphyton, and macrophyte data from the Rotenberger Wildlife Management Area, 1998. South Florida Water Management District, West Palm Beach, Florida.

Tomkins IB, Kellas JD, Tolhurst KG & Oswin DA (1991) Effects of fire intensity on soil chemistry in a eucalypt forest. Aust. J. Soil Res. 29: 25-47.

Turner, C.L., J.M. Blair, R.J. Schartz and J.C. Neel (1997) Soil N and Plant Responses to Fire, Topography and Supplemental N in Tallgrass Prairie. Ecology 78: 1832-1843.

United States Army Corps of Engineers (1986). Test Methods for Evaluating Solid Waste, Physical and Chemical Methods, 3rd edition.

United States Environmental Protection Agency (1983) Methods for the chemical analysis of water and wastes. EPA-600/4-79-020, revised. Office of Research and Development, United States Environmental Protection Agency, Cincinnati, Ohio.

Vogl RJ (1969) One hundred and thirty years of plant succession in a Wisconsin lowland. Ecology 50: 248-255.

Vogl RJ (1973) Effects of fire on the plants and animals of a Florida wetland. Am. Mid. Nat. 89(2): 334-347

van Arman J & Goodrick R (1979) Effects of fire on a Kissimmee River marsh. Florida Sci. 42(4): 183-194.

Volk BG, Schemnitz SD, Gamble JF & Sartain JB (1975) Base-line data on Everglades soil-plant systems: Elemental composition, biomass, and soil depth. In: Howell GF, Gentry JB & Smith MH (Eds) Mineral cycling in Southeastern ecosystems (pp 658-672). National Technical Information Service, Springfield, VA.

Wade DJ, Ewel J & Hofstetter R (1980) Fire in South Florida ecosystems. Unites States Forest Service General Technical report no. SE-17. Asheville, N.C.

Weiss TE (1980) The effects of nutrient availability on the pitcher plant Sarracenia flava L. Ph.D. Dissertation. University of Georgia, Athens. 108 pp.

Wilbur RB & Christensen NL (1983) Effects of fire on nutrient availability in a North Carolina coastal plain pocosin. Am. Mid. Nat. 110: 54-61.

Table 1. Constituent:TCa ratios in soils (0-2 cm layer) from pre-fire muck-burn (PRE_{MB}), post-fire muck-burn ($POST_{MB}$) pre-fire surface-burn (PRE_{SB}), and post-fire surface-burn ($POST_{SB}$) areas.

	TP	TN	TC	%Ash	KCl-SRP	HCl-SRP	NaOH-SRP	NaOH-Po	total iP
PRE_{MB}	0.018	1.13	15.28	22.36	0.07	5.97	2.73	15.10	8.77
$POST_{MB}$	0.014	0.31	3.55	14.16	0.06	18.50	0.76	1.36	19.33
PRE_{SB}	0.022	0.86	11.94	12.95	0.03	7.54	2.57	11.92	10.14
$POST_{SB}$	0.018	0.80	10.71	17.64	0.18	9.38	2.00	6.37	11.55

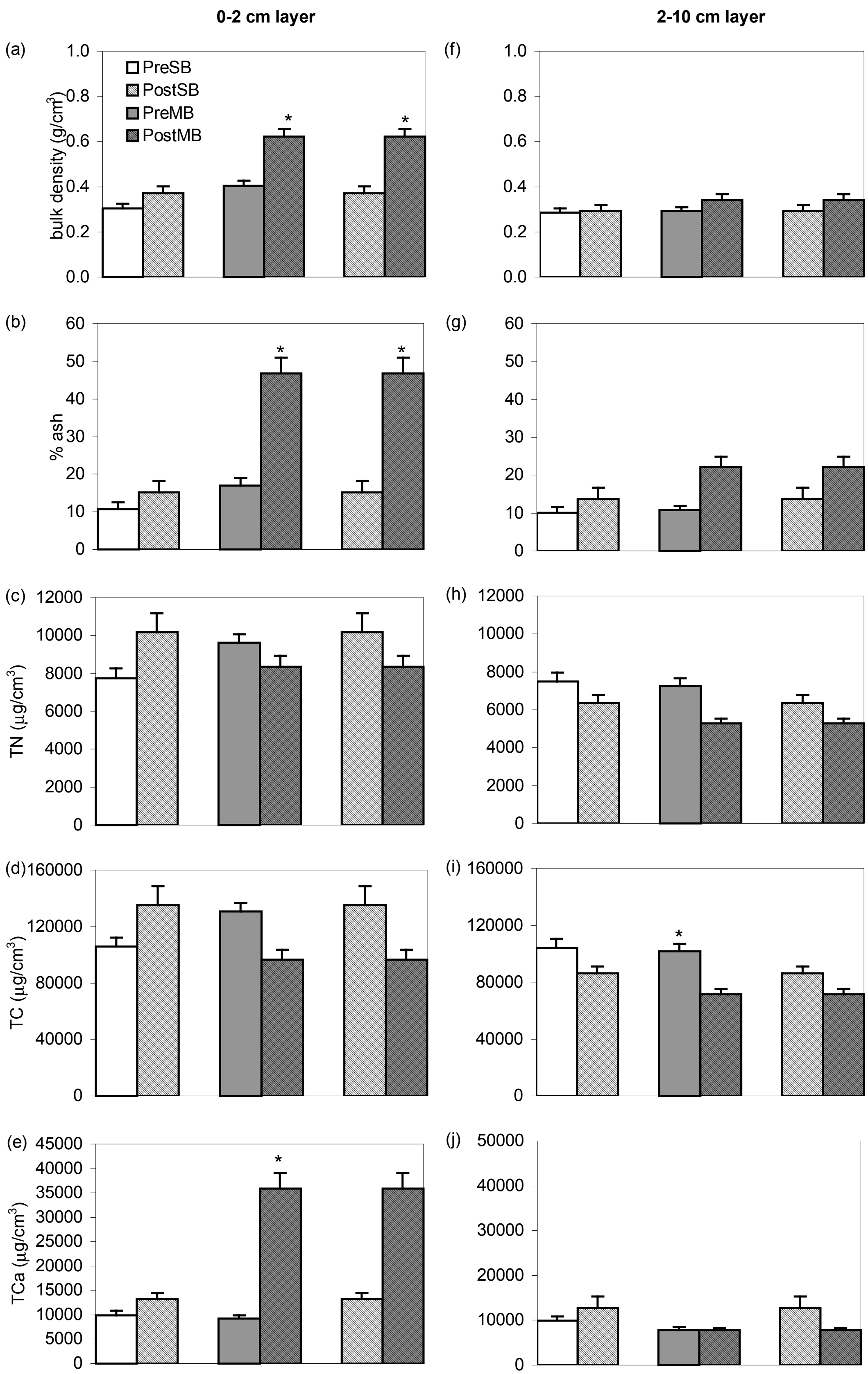
Figure legend.

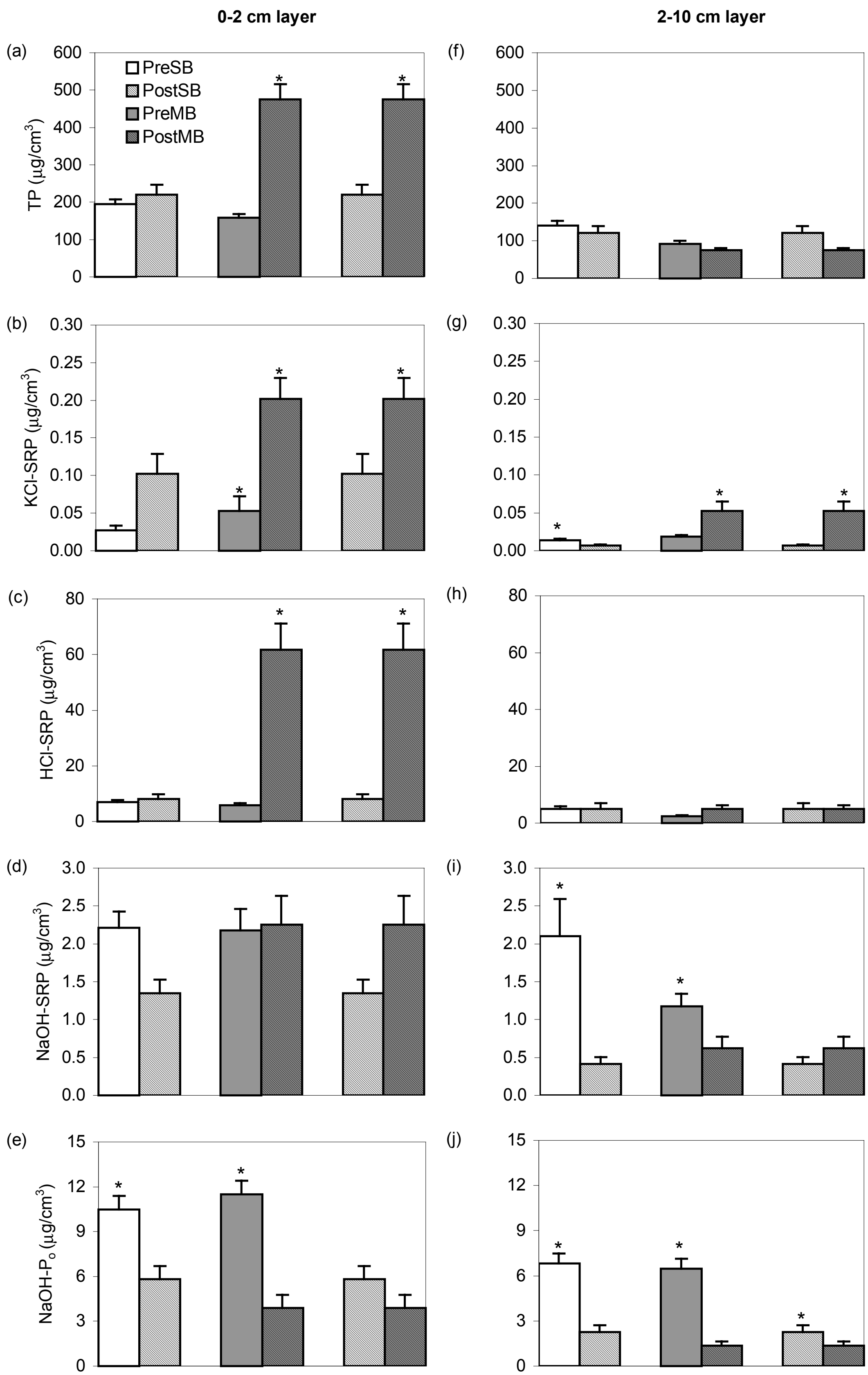
Figure 1. Map of South Florida showing the Rotenberger Wildlife Management Area (RWMA) and sampling locations (closed circles) (major canals and levees of South Florida's water management system are included).

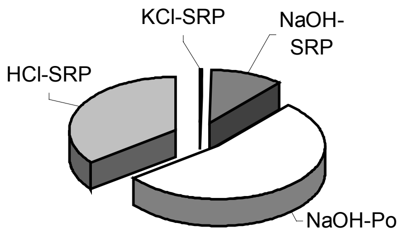
Figure 2. Comparisons of bulk density (BD), ash content (% ash), total nitrogen (TN), total carbon (TC), and total calcium (TCa) in the 0-2 and 2-10 cm soil layers for selected pairs of treatments (bars are means + 1 standard error; asterisks denote significant difference between treatment pairs).

Figure 3. Comparisons of total P and P fractions in the 0-2 and 2-10 cm soil layers by burn type (bars are means + 1 standard error; asterisks denote significant difference between adjacent treatment means).

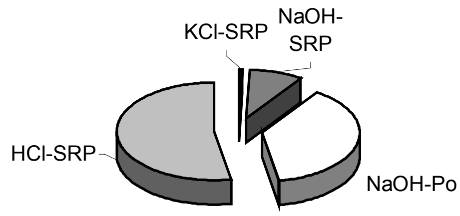
Figure 4. Proportions of inorganic and organic P fractions in soils (0-2 cm layer) from pre-fire muck-burn (PRE_{MB}), post-fire muck-burn ($POST_{MB}$) pre-fire surface-burn (PRE_{SB}), and post-fire surface-burn ($POST_{SB}$) areas.


0-2 cm layer
2-10 cm layer
(a)
(b)
(c)
(d)
(e)
(f)
(g)
(h)
(i)
(j)
PreSB
PostSB
PreMB
PostMB
bulk density (g/cm3)
% ash
TN (µg/cm3)
TC (µg/cm3)
TCa (µg/cm3)

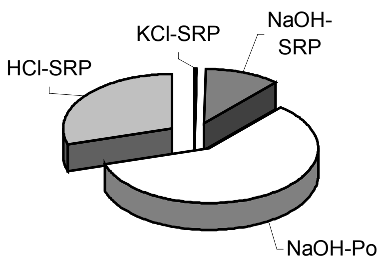

0-2 cm layer
2-10 cm layer
(a)
(b)
(c)
(d)
(e)
(f)
(g)
(h)
(i)
(j)
PreSB
PostSB
PreMB
PostMB
TP (µg/cm3)
KCl-SRP (µg/cm3)
HCl-SRP (µg/cm3)
NaOH-SRP (µg/cm3)
NaOH-Po (µg/cm3)



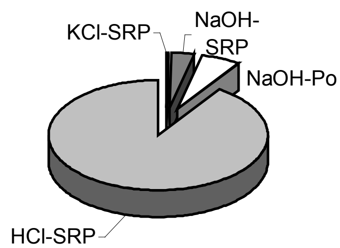
PRE-SB



POST-SB



PRE-MB



POST-MB